# Report of the Working Group on Seabird Ecology (WGSE)

## 19–23 March 2007

## Barcelona, Spain

**International Council for the Exploration of the Sea**
**Conseil International pour l'Exploration de la Mer**

# Contents

# 1 Executive summary

## Highlights

- WGSE has summarized and commented on the current approaches on how to identify and delineate protected areas for seabirds. This is a burning issue as many protected areas according to the EU Birds Directive have still to be designated in marine areas, especially in offshore areas. The concepts for Special Protection Areas (SPAs), following the EU Birds Directive, and Important Birds Areas (IBAs), according to Birdlife International, are explained and their applications are demonstrated by several case studies, as are indications for future work needed.

- A concept for a comprehensive seabird monitoring programm were developed further. New information could be included from an international workshop on seabird population monitoring in Europe and from recent results of the Scottish Seabird Monitoring Programm. A new section on the use of seabirds as bio-monitors of change in the marine environment has been incorporated.

- WGSE has reviewed how much evidence of climate-related changes exists for seabirds in the OSPAR Maritime Region. Parameters involved are changes in distribution, reproductive success, survivorship and different phenological aspects.

The Working Group on Seabird Ecology (WGSE) met for five days (19–23 March 2007), and was attended by fifteen persons from eight countries (Annex 1). Twelve were nominated members of the group and three were nominated by the Working Group Chair to attend this year's meeting. During the meeting WGSE was able to address all Terms of Reference and the results are reported here.

The chapter reviewing the current approaches how to identify and delineate protected areas for seabirds was finalised. This topic has become an important issue during recent years as most ICES member states are involved in designating Special Protection Areas (SPAs), following the EU Birds Directive, and Important Birds Areas (IBAs), according to Birdlife International. The different concepts are explained and their applications are also demonstrated by several case studies, as are indications for future work needed.

Progress was also made on the chapter related to monitoring seabirds. In addition to the work presented last year, three issues could now be included: (1) results of a workshop on seabird population monitoring in Europe, (2) recent results of the Scottish Seabird monitoring with findings of a recent analysis of productivity data and (3) a new section on the use of seabirds as bio-monitors of change in the marine environment.

It was possible to finalise the chapter on how diet in seabirds can be sampled and should be sampled according to a variety of parameters. Also, recommendations are given how to report information from dietary studies so that the data can be used best.

As a new topic for WGSE, ecological issues related to the circulation of pathogens and parasites in seabird populations were reviewed. This chapter presents a first attempt on the topic, which includes a general outline of the subject and some details on specific issues.

OSPAR has requested ICES (and thus WGSE) to assess the changes in the distribution and abundance of marine species in the OSPAR maritime area in relation to hydrodynamics and sea temperature. In this chapter, an overview is given of how much evidence of climate-related changes exists for seabirds in the OSPAR Maritime Region. Parameters involved are changes in distribution, reproductive success, survivorship and different phenological aspects.

As another OSPAR request, WGSE was asked to review nine nominations for the OSPAR list of threatened and/or declining species. Comments on five of the nominated species could be

given before the meeting while comments on the other four species were finalised during the WGSE meeting. Recommendations were given as to whether sufficient evidence of declines/threats exists and comments were given as to whether the reviews were sufficiently complete.

## 2    Introduction

### 2.1    Participation

The following members of the Working Group on Seabird Ecology (WGSE) participated in the meeting (see Annex 1 for full details).

| | |
|---|---|
| Tycho Anker-Nilssen | Norway |
| Pep (J. M.) Arcos | Spain |
| Rob Barrett | Norway |
| Thierry Boulinier | France |
| John Chardine | Canada |
| Bob Furness | UK |
| Stefan Garthe (Chair) | Germany |
| Ommo Hüppop | Germany |
| Bill Montevecchi | Canada |
| Ib Krag Petersen | Denmark |
| Iván Ramírez | Portugal |
| Norman Ratcliffe | UK |
| Jim Reid | UK |
| Richard Veit | USA |
| Carlota Viada | Spain |

Twelve persons were nominated members of the group; three persons were nominated by the WG Chair to attend this year's meeting. The possibility to nominate persons not yet nominated by national delegates was again considered by the group to be an extremely useful tool.

### 2.2    Terms of Reference

The 2006 Statutory meeting of ICES gave the Working Group on Seabird Ecology [WGSE] the following Terms of Reference:

a )   finalize reviewing the current approaches for identifying offshore seabird aggregations and delineating Important Bird Areas (IBAs) and Special Protection Areas (SPAs);

b )   continue developing recommendations for a comprehensive monitoring programme for seabirds;

c )   finalise reviewing on how to sample diet and how to report results of dietary studies in seabirds, and develop recommendations for future field studies and analyses;

d )   consider scientific ecological issues linked to the circulation of parasites and pathogens within seabird populations;

e )   assess and report on changes in the distribution, population abundance and condition of seabirds in the OSPAR maritime area in relation to changes in hydrodynamics and sea temperature;

f )   assess and report on the extent to which the changes reported in (e) can reliably be attributed to changes in hydrodynamics and sea temperature;

g )   assess and report on the evidence on which the nominations of Arctic loon *Gavia arctica*, Balearic shearwater *Puffinus mauretanicus*, Band-rumped storm-petrel *Oceanodroma castro*, European shag *Phalacrocorax aristotelis*, Greater scaup *Aythya marila*, White-winged scoter *Mellanitta fusca*, Black-legged kittiwake *Rissa tridactyla*, Ivory gull *Pagophila eburnea* and Thick-billed murre *Uria lomvia* for the OSPAR List of Threatened and/or Declining Species and Habitats are based. The purpose of each assessment is to ensure that the data used to support each nomination are sufficiently reliable and adequate to serve as a basis

for conclusions that these species can be identified as threatened and/or declining species according to OSPAR's Texel/Faial criteria

## 2.3 Note on bird names

Throughout the text we provide common English names for bird species. In Annex 3 a full list of both English and scientific names is given.

## 2.4 Acknowledgements

The Working Group wishes to thank SEO/BirdLife for the organisation of the logistics in Barcelona, and to the Museu de Ciències Naturals de Barcelona for providing us with meeting rooms and other facilities. Camphuysen, C. J., Fox, A. D., M. Frederiksen, Leopold, M. F., A. Mosbech, M. Nunes, D. Oro, A. Petersen, S. Pihl and M. L. Tasker provided information that was very useful for this report.

# 3 Current approaches for identifying seabird aggregations and delineating Important Bird Areas (IBAs) and Special Protection Areas (SPAs) at sea

The identification of Important Bird Areas (IBAs) and Special Protection Areas (SPAs) at sea is assuming increasing importance in the protection of birds in the marine environment in Europe and elsewhere. The application of these conservation tools to marine areas has lagged behind that in terrestrial habitats for a variety of reasons but their potential utility here is certainly no less important. In this chapter, we review the various issues that are of most relevance for the establishment of IBAs and SPAs networks at sea.

## 3.1 Introduction

### 3.1.1 Definitions and rationale for site identification

There are several types of potential protected area for a variety of marine system components. In the marine environment, the abbreviation MPA, if/when used here, merely denotes any marine protected area. Very often, however, a MPA would refer to an area at sea that is protected primarily for habitats or fish stocks. The principal instrument in the European Union for the classification of important habitats is the Habitats and Species Directive (EC, 1992). This piece of legislation provides not only for the designation of sites for important habitats for the whole range of organisms including birds, but also sites for most taxa. Such sites are known as Special Areas of Conservation (SAC). A related piece of legislation, the Birds Directive (EEC, 1979), provides for the protection of birds. So, under the Birds Directive, a Special Protected Area (SPA) is an area specifically classified for its important bird interest. Together, SPAs and SACs are known as Natura 2000 sites. There may be political as well as practical conservation benefits to identifying sites that qualify as both SPAs and SACs.

An Important Bird Area (IBA) also denotes an area that hosts birds in significant numbers according to specific thresholds and criteria (though not exclusively for birds that form aggregations – widely dispersed species may also be the focus of IBA identification). There are notable differences between the statutory designation SPA and the accolade IBA.

Articles 4.1 and 4.2 of the Birds Directive require that Member States classify "the most suitable territories in number and size as special protection areas" for those bird species included in Annex I of the Directive and also for regularly occurring migratory species of bird, taking account of their protection requirements at sea as well as on land. In this context, "regularity" is usually defined according to the Ramsar Convention (www.ramsar.org/) – that the requisite number of birds is known to have occurred in two thirds of the seasons for which adequate data are available (the total number of seasons being not less than three) or that the mean of the maximum count of those seasons in which the site achieves international importance (taken over at least five years) reaches the relevant threshold. The SPA concept is a legally binding one; failure to classify SPAs for important bird populations by Member States of the EU carries the risk of legal proceedings being instituted against those Member States. SPAs are defined by applying criteria and guidelines formulated by state governments or their advisors.

Although Articles 4.1 and 4.2 place an absolute obligation on Member States to classify areas as SPAs for Annex I and migratory species, it also requires the application of other special conservation measures in respected of these species. That is, special conservation measures for these species are to be taken in addition to classifying SPAs – SPAs are not deemed by themselves to constitute the special conservation measures. The species included in Annex I require special protection in the European Union because they are rare, in danger of extinction, vulnerable to specific changes in their habitat or because they require particular attention for other reasons (CEC 2006). EU Member States may apply some discretion in the

choice of SPAs, but their classification and delimitation must be based only on the ornithological criteria determined by the Birds Directive; socio-economic factors may play no role in their classification (CEC, 2006).

The concept of an Important Bird Area applies globally rather than just within Europe. IBAs carry no legal weight. Areas are accorded IBA status by using criteria compiled by BirdLife International partners. However, this does not weaken the scientific merit of the process. Indeed, the European Commission takes the view that the list of European IBAs compiled in 2000 identifies at least some of those areas deemed most suitable for the conservation of relevant species. Furthermore, the fact that lists of IBAs exist for Member States that have made little progress in identifying potential SPAs is indicative of those Member States having failed to meet their legal obligation to classify SPAs, and this has been repeatedly confirmed by different European Court rulings asking governments to use their IBA inventories (RSPB, 2004). IBA lists therefore comprise at the very least, some areas that would qualify as SPAs.

Though not identical and founded on different criteria (see Section 3.3), there may be in practice a great deal of overlap between SPAs and IBAs, certainly in the broad areas so identified if not their exact boundaries (for example, RSPB, 2005). Such an overlap is reinforced at a European level where a specific category of the IBA criteria was adapted to the EU bird populations (category C, see Section 3.3). Both these types of protected area aim to protect discrete concentrations of birds. What constitutes a discrete concentration as opposed to a widely dispersed pattern of birds is a question of scale, and ultimately the distinction to be made becomes a subjective judgement. However, that is not to say that concentrations may not be identified using "objective" and repeatable methods that result in operational definitions that can be applied in consistent ways (see Section 3.5). The important point is that the protection requirements of species must be addressed. Depending on the nature of species dispersion it is sometimes important to fulfil these requirements via site designation and sometimes through wider conservation measures. It is desirable to combine site-specific, species-specific and activity-specific approaches to guarantee the protection of species; however, it is a legal requirement on Member States of the EU to classify SPAs as part of a wider suite of protection measures for those species listed on Annex I of the Birds Directive and also for regularly occurring migratory species.

### 3.1.2    General principles in identifying important marine areas for birds

Compilation of a network of SPAs in the marine environment has reached only a relatively early stage across Europe; the focus over the past three decades has been firmly on terrestrial, freshwater and inter-tidal sites. In the absence of prescriptive advice from the EU, rules and guidelines for the identification and classification of SPAs across the European Union have varied across Member States. However, many states have adopted similar approaches. Given that the process is now fairly well established for terrestrial/freshwater/inter-tidal habitats over most of the EU, and also for the reasons of consistency, there has been a presumption that those guidelines formulated for non-marine environments should, as far as possible, be applied to the marine environment. Hence, little distinction has been made between the two environments, though simple transposition of the terrestrial model into marine areas is not without its difficulties (see Section 3.1.3).

Guidelines that have been applied to determining whether sites qualify as SPAs, certainly in those EU Member States that have made significant progress in compiling a suite of SPAs, tend to specify that specific proportions or numbers of relevant populations be represented within the sites. For example, areas that are used regularly by 1% or more of the national or biogeographical populations of species in any season, or that host particularly high densities, or contain more than 20 000 waterfowl or seabirds in any season, are deemed as meriting classification as SPAs. Notwithstanding this, there always remains scope for applying judgements based on other sorts of (reliable) information on species' ecology and life

histories. More information on SPA qualification guidelines in some EU Member States is provided in Section 3.6.

In contrast to SPAs, the criteria used to assess whether a site qualifies as an IBA are universally applicable. Some of these criteria are expressed again in terms of 1% thresholds of relevant populations including flyways, and numbers of individuals (with 20 000 again being the preference). There is also a significant emphasis on the perceived threat or conservation status of the species, so special criteria for globally endangered species have also been established. In common with the SPA issue, the IBA criteria were devised specifically with regard to bird populations inhabiting terrestrial, freshwater and intertidal habitats. In fact, IBAs may include coastal waters to 20 m depth; Tucker and Evans (1997) identify 470 IBAs in Europe that include some marine habitat, 147 of which have more than 50% marine habitat cover and at least 25 of which are wholly marine. However, marine IBA criteria are under review (see Section 3.3) and there is recognition that they should consider the whole marine environment and extend from low water rather than from the somewhat arbitrary 20 m isobath.

It will be clear that while the two concepts are very similar and share common purposes, SPAs and IBAs differ in their respective philosophical provenances. Both the IBA and SPA (or networks thereof) identification processes will be tempered by political considerations, on the one hand by non-government organisations whose principal remit will be the strict protection of birds and their habitats, and on the other by state governments whose objectives might be influenced by wider political and socio-economic concerns. These latter concerns do exist and the potential/perceived regulatory impact of designating a suite of SPAs might be real. However, the identification of sites that qualify as SPAs, and determination of their limits, must be done by applying sound scientific practice using the best possible data available. The same considerations must prevail in identifying IBAs.

Notwithstanding the EU Guidelines for implementation of the Habitats and Birds Directives in the marine environment (EU 2006), no generic, prescriptive rules exist across the EU for the identification of SPAs. The responsibility for identifying SPAs rests with Member State governments, though IBA lists may aid in this. While consistency of approach across the EU might be desirable for many reasons, it is perhaps not possible either practically or politically.

### 3.1.3  General difficulties in identifying important marine areas for birds

Although IBA and SPA selection criteria have been developed for all biomes, most guidance has derived from previous work in the terrestrial and coastal zones, with the latter also usually been viewed from a land perspective. Consequently, there are specific problems when considering important areas for birds at sea.

Information on distribution and abundance of birds at sea is not as good as that for birds on land. Studies of seabirds at sea started late compared to most habitats and areas on land. Substantial progress has been made in the North Atlantic over the past three decades however, and many marine regions are now quite well known (the North Sea for example), and several countries have begun research and surveys aimed at collecting information to identify protected areas (for example, the UK, Spain, Portugal, Malta and the Baltic countries). However, seasonal constraints tend to result in more bird surveys being conducted in summer than in winter months and coverage varies according to distance from shore. Many areas (at small scales of analysis) have been surveyed infrequently if at all, and therefore represent "snapshots" of marine bird distribution and abundance. This means it is difficult to determine whether sites are used consistently or ephemerally. Furthermore, sea areas are vast and remote from land and thus there are hardly any options to sufficiently cover such areas even once. The Macaronesian Sea and the northwest Atlantic are two such examples.

The activity range of seabirds – although very variable – tends to be much larger than on land, even if compared to birds of prey that have rather huge territories. Chiefly, the extent of spatial use of the seas is related to foraging activities and migration. In contrast to most birds that feed on land, foraging ranges of breeding seabirds may extend as far as several hundred kilometres from the colonies as in the case of most Procellariiforms (Shealer, 2002). Consequently, it will be challenging to identify areas of higher intensity of use for inclusion in any IBA/SPA. However, many species have much shorter foraging ranges, especially terns and auks (e.g. Pearson, 1968; Garthe, 1997). For wide-ranging species, most foraging areas may very likely be disconnected from the seabirds' colonies, thus requiring independent protection.

Seabirds generally have dispersed distributions but most species associate with habitat features. However, such habitats are often comparatively large-scale and the site boundaries rather indistinct and often also mobile compared to terrestrial, freshwater and intertidal areas. This results in relatively low spatial stability in seabird concentrations, which makes site-based conservation difficult, especially at smaller scales. However, there may be ways incorporating this in hydrographic models (see Section 3.5.3).

## 3.2   Marine protected areas for birds

There are several types of potential protected areas for birds in the marine environment. Broadly these may be categorised as:

- extensions of existing terrestrial protected areas such as seabird breeding colonies into the adjacent sea. Such extensions of SPAs or IBAs could be for feeding or other purposes, principally during the breeding season;
- "inshore" areas used by concentrations of species such as divers and seaduck usually, but not necessarily, outside the breeding season. Although termed "inshore" such concentrations could extend quite far from the coast, beyond 12 nm for example;
- "offshore" areas used by concentrations of seabirds. Again, such areas need not be confined to those at great distances from the coast; seabirds do not respect such arbitrary boundaries as 12 nm (territorial limits) or 200 nm (EEZ limit). Large numbers of seabirds might congregate in certain areas mainly because of the abundance of food, but other factors might determine this such as the need for shelter, or hydrographical features; and
- migration bottlenecks or hotspots. Such areas aimed at protecting migrating birds refer to any area of open sea over which significant concentrations fly and whose use may be variable over time. These areas may also include staging areas where birds regularly congregate in significant concentrations during migration.

To reiterate, the distinction being made between offshore and inshore concentrations here is rather arbitrary. The latter refers principally to an approach that considers certain species that mainly congregate close to the coast outside the breeding season – mainly seaduck and divers, but also perhaps some gulls and grebes; other seabirds that occur close to the shore will be captured by this approach but will also be included in the offshore approach. The latter addresses the occurrence of all birds for which data exist in the marine environment. "Inshore" concentrations would largely be surveyed, for logistical and analytical reasons, using aircraft, whereas "offshore" distributions of birds would be determined mainly by ship-based surveys. Of course, there will be overlap between the two and all types of SPA or IBA have the potential to be contiguous with each other. BirdLife International is currently proposing a new classification that will join these two types of concentration (SPEA-SEO/BirdLife, 2006).

### 3.3    Guidelines and criteria for determining SPAs and IBAs

As mentioned earlier, there is no EU agreed set of criteria that may be applied in defining SPAs. This is in marked contrast to IBAs, where globally applicable criteria have been (or are being) formulated. SPA guidelines have been formulated by several countries within the European Union and many of these are very similar; they are summarised in the relevant country sections below. Many involve the application of numerical or density thresholds to determine SPA qualification as well as further judgements based on other ecological considerations such as relative population size or density, species range, breeding success etc.

The approach to identifying IBAs combines the threshold and judgement stages of the SPA process (BirdLife, 2005). IBA criteria were developed specifically for the selection of IBAs in Europe (divided into three main categories, A Globally, B European and C EU). These allow the identification, based on a (marine) site's international importance for:

- threatened bird species, according to IUCN and SPEC categories (criteria have been defined for A, B and C categories); and
- congregatory bird species (criteria have been defined for A, B and C categories).

The definition of IBA networks in the marine environment has so far been focussed mainly in coastal waters out to the 20 m isobath. The major gap, offshore waters, is currently being addressed by SEO/BirdLife and SPEA, and should be done in a way that considers the whole marine environment beyond low water.

While the criteria developed for terrestrial IBAs for threatened bird species can be fairly easily transposed to the marine environment, the main difficulties appear with the set of criteria for congregatory bird species. This may render their populations particularly vulnerable because of potential destruction or degradation of those sites at which they congregate when breeding, wintering or on passage. Consequently, there are some important considerations to be made when adapt the existing IBA criteria for application to the marine environment:

- the congregatory criteria were originally formulated for waterfowl by the Ramsar Convention and are not wholly pertinent to seabirds;
- most pelagic seabird species do not always congregate at feeding areas in the same way that waterfowl do. Numerical thresholds to qualify areas as IBAs for feeding seabirds must be adapted in order to accommodate intense use of a site by numbers that do not necessarily reach at a given time, for example, 1% of population thresholds;
- given the difficulties in surveying large marine areas, environmental variables could help in defining the distribution patterns of the most pelagic species;
- many seabird species do migrate, even well offshore, across fairly narrow corridors. The IBA criteria should therefore be adapted to apply to the specific areas where bird passage is distinctly intense or is at particularly risk. The stopover sites, where the species spend several days feeding and resting may then be identified as IBAs applying the congregatory criteria for bottleneck sites;
- "transit corridors" that seabirds regularly use in small scale movements such as between colonies and the feeding areas could be considered for IBA identification. The intensity of use can be very high;
- The 1% threshold that set up the numerical thresholds in some of the BirdLife criteria in the three different categories (A, B and C), is useful for identifying IBAs for waterfowl but it seems that could create problems to ensure the minimum coverage of the most important pelagic seabird areas.

These are the most important issues that need to be addressed when attempting to apply terrestrial IBA criteria to pelagic seabird distributions.

## 3.4    Methods of data collection

Recognition of IBAs and SPAs depends on identifying sites where birds occur and the numbers of birds associated with these, and this requires data. Data can be collected specifically to identify SPAs or IBAs or data on distribution and abundance from other sources (e.g. Environmental Impact Assessments for windfarm or oil developments) can be collated and used for this purpose. Data collected specifically for protected area purposes have obvious advantages, since scales and coverage surveys can be designed to address site determination. However, coverage and scales from other surveys can be adequate for this purpose, in which case lengthy and expensive survey work can be avoided. In some cases, gaps may exist in the available data; these may be filled by dedicated survey, in which case the two sources of data are complementary.

As indicated by much of the above, there are two broad methods of collecting such data on seabird distribution, and these are described in the following sections.

### 3.4.1    Transect survey

Open waters are too extensive to allow complete counts of birds inhabiting them, such that a sampling of densities followed by extrapolation has to be employed. Transect counts are the most common means of sampling seabird density at sea. This provides data on the population scale, with the distribution of a large proportion of the population being described if the survey area is adequately wide and resolution sufficiently fine.

There are two main types of transect surveys. Strip transects use a fixed transect width, within which the aim is to detect all individuals. A strip width of 300 m is the often used. Line transect surveys use observations from a wider transect width that is subdivided into distance bands to estimate densities. The decreasing detection probability with increased distance away from the survey track line is used to fit a detection function using Distance Sampling (Buckland *et al.,* 2001). A crucial assumption is that all birds are detected on the innermost transect line. Transect data are usually resolved into birds per km$^2$, and presented as density grids or contour plots.

Counts can be made from ships or aircraft; an evaluation of the two platforms can be found in Camphuysen *et al.* (2004). Aerial surveys allow rapid coverage of large survey areas and access to shallow areas or complex coastlines, whereas boat surveys are more suitable for offshore areas or restricted waters. Identification and detection of cryptic species (auks, storm-petrels) is more difficult from aircraft than from a boat, although easily flushed species (such as divers and seaduck) may flee from slow moving boats before they are counted whereas fast-moving planes are able to detect them as they take flight.

The advantages of transect surveys are that they are able to sample distribution of a large proportion of a population if coverage is sufficiently wide, and that estimates of numbers within areas can be tentatively calculated. The main problem with transect surveys is that they only provide information of distribution within the area covered, and where coverage is incomplete biases in assessment of distribution and relative importance of areas will result. However, spatial modelling using environmental covariates can reduce this problem, provided that sufficient explanatory variables can be identified (Christensen *et al.,* 2006; Petersen *et al*., 2006). Furthermore, the provenance of birds is unknown, which is problematic when assessing the importance of areas for birds from particular colonies or populations. Similarly, transect surveys fail to provide information on the age class (only for some species) and the breeding status of the birds observed. Finally, as the technique is visual, no data are obtained at night and distributions of some birds may exhibit diurnal variation.

### 3.4.2 Tracking individual birds

Tracking involves fitting devices to individual seabirds that store or transmit data that can be used to determine their locations at sea at varying time intervals. A general rule applies that the device should not be heavier than 5% of the bird's body mass (Cochran, 1980) which restricts it use on small species. A variety of devices are available for this purpose and these are described below.

Radio-tracking involves fitting birds with a radio-transmitter, and the signals from this can then be detected by an antenna. This allows the location of a bird to be determined by triangulation from fixed points (Freeman *et al.,* 1997; McSorley *et al.,* 2005, see above) or by following them by boat (Ostrand *et al.,* 1998) or plane (Adams *et al.,* 2004; Mañosa *et al.,* 2004). Radio-tracking is relatively cheap, and transmitters can weigh less than 1g, such that tags can be fitted to even the smallest seabirds. However, detection range is often limited to an order of tens of km, depending on transmitter size and height of the receiving antenna. As such, they are only suitable for determining foraging range of relatively inshore species unless individuals are followed intensively by an aircraft, or rafting areas of pelagic species (McSorley *et al.,* 2005) around colonies. Attempts to use radio-tags to track foraging trips small petrels in Macaronesia were largely unsuccessful, despite being combined with light aircraft to improve effective detection range (see below).

Platform Terminal Transmitters (PTTs, also known as satellite tags) transmit position data regularly to orbiting satellites and hence to the observer and so can be detected at any point of the globe without the need for retrieval, allowing wide-ranging, pelagic seabirds to be tracked. Accuracy is relatively high, with an error of usually few km at most (Wilson *et al.,* 2002). Until recently, PTTs were heavy and bulky and so could only be fitted to large birds such as albatrosses (Jouventin and Weimerskirch, 1990) and penguins (Davis and Miller, 1992). The size of these devices has decreased substantially in recent years (minimum 9g to date), which allows deployment on medium-sized species such as some shearwaters. However, PTTs are still too heavy for small species such as terns and small petrels.

GPS loggers calculate positions from to orbiting satellites and store these. They provide the highest accuracy available, but have to be retrieved to download the data. Size constrains have limited their use to large species such as albatrosses (Weimerskirch *et al.*, 2002) and gannets (Camphuysen, 2005, Garthe *et al.*, 2007), but improvements in power of storage and reduced size could make GPS loggers very useful tracking devices in the next few years. Combined GPS-PTTs now weigh as little as 21g and merge the high accuracy of the GPS system and do not need to be recovered to obtain the data.

Global location loggers (geolocation loggers, light loggers) carry an internal clock and register light intensity, from which daylight duration and sunrise and sunset hours, and hence latitude and longitude, can be calculated. This technique is relatively cheap and easy to conduct, though it is necessary to recapture the tagged birds to download the stored information. Precision is low (tens of km), and it is best recommended to employ these loggers for wide-ranging species and also for wintering behaviour rather than foraging area identification while breeding.

Compass loggers have two or three compasses along with an internal clock. From the heading of the compasses and the flight duration a paths can be reconstructed. Flight routes of species exhibiting relatively straight flights and not showing too many changes in activity (e.g. gannets) are much easier to reconstruct than those from species turning very often and changing activity very frequently (e.g. kittiwakes and shearwaters). Short time intervals between data logging increase the number of points registered and the accuracy of the flight route.

The advantage of tracking is that it can provide accurate data that covers a seabird's global range that is not bounded by arbitrary survey areas (as transect surveys may be). Remote-sensing techniques also allow the provenance of birds in different areas to be assessed and reveal information on seabird movements during the night. The disadvantages are that some tags are still expensive and so the number of birds and colonies at which they can be deployed is limited. As such, distribution may not be representative of the population as a whole owing to variation in ranging behaviour according to colony, age, sex, breeding status, individual and season.

Transect methods and tracking can be complementary with both methods providing independent data on distribution, tracking revealing provenance of birds and transect counts providing estimates of numbers (Camphuysen *et al.*, 2004). Table 3.1 compares the two approaches.

**Table 3.1. Comparison between transect surveys and individual bird tracking.**

|  | Transect surveys | Bird tracking |
|---|---|---|
| Approach | Population | Individual |
| Representativity of population | Good (potential biases) | Restricted to a fraction of the population (e.g. breeding birds) |
| Bird information | Very limited | Good |
| Behaviour and interactions with environment | Direct information | Inferred information |
| Spatial coverage | Good (limited by survey constrains) | Good (within range of species and technical limitations of the devices) |
| Temporal coverage | Limited (or high effort required) | Good (limited by device life-span) |
| Activity rhythms | Lack of nocturnal data | Day-round information |
| Cost | Low (if "opportunistic" base) | Medium to High |
| Sample size | High | Low |
| Potential biases due to: | Temporal changes (at different scales) Lack of any information of the birds observed: breeders vs. non-breeders, colony location, etc. Type of boat used (commercial, research vessel etc) | Significant fraction of population disregarded Sample size Device autonomy and accuracy |

## 3.5   Determination of protected area boundaries

Once areas have been identified as hosting important numbers or densities of priority, bird species boundaries to the specific sites to be accorded protection, need to be determined. There are many ways of defining site boundaries, depending on the nature of the information and data available, and on the type of SPA or IBA that is deemed appropriate. The task is clearly more difficult for areas and species for which relatively little information exists, especially at finer scales.

### 3.5.1   Colony extensions

In order for an IBA or SPA protected colony to be extended, clearly it must be classified as such in the first place. The size of the colony extension then needs to be made in the context of the bird's use of the adjacent waters rather than in any population size context. This could be an area within which a desired proportion of the activity of interest occurs. McSorley *et al.* (2005) assessed the use made of adjacent waters of several species of seabird around a sample of colonies in the UK. They categorised seabird activity into site-specific (for example foraging) and non site-specific activities (such as maintenance behaviours). The distributions of those birds around the colonies engaged in non site-specific behaviour were used to inform the determination of generic colony extensions for the relevant species at all SPA colonies where they occur. This solution of course does not capture all of the foraging range of birds at

colonies. Different solutions are required here and Birdlife (2005) has proposed protected areas based on the typical foraging ranges of different species. These are presented in Table 3.2. For some species, such generic proposals may not be appropriate. Manx shearwater SPA colony extensions have been addressed on an individual colony basis in the UK.

Those colony extensions already determined have derived from survey data collected by various means, including ship-based at-sea surveys and radiotracking. Possible feeding extensions for terns will also be explored using aerial surveys. Other methods are appropriate, and work is under way investigating breeding site extensions for red-throated divers and shags using habitat modelling methods.

Colonial nesting seabirds are central place foragers, and many species are highly aggregated in discrete colonies. As such, the distribution of breeding seabirds tends to be more clumped and spatially stable during the breeding season than at other times of year. Furthermore, many of the most important seabird colonies are already recognised as IBAs and designated as SPAs, but these generally only extend to the high water mark. Marine extensions to such colonies may therefore have merit for protecting bathing, resting or foraging areas upon which birds breeding at the colony depend for survival and successful reproduction, and various approaches exist to delimit the boundaries of these.

Bathing and resting birds often congregate around cliff colonies at high densities that can be incorporated into the colony SPA using generic species-dependent extensions. Ship-based surveys of seabird distribution revealed densities of auks declined markedly at 1 km from the colony and those of northern fulmar and northern gannet at 2 km from the colony, and these limits were used to define boundaries for marine extensions to colony SPAs (McSorley *et al*., 2004). The seaward boundary of Manx shearwater rafts around three UK colony SPAs was determined from fixes of radio-tagged birds (McSorley *et al*., 2005; Box 3–1). However, most species feed beyond the boundaries of resting areas and such extensions will not recognise or protect important foraging areas.

The boundaries of foraging areas can be estimated for some species by using foraging radii (Birdlife International, 2000). These can be determined from empirical observations of foraging ranges using tracking devices or transect surveys from ships as described in Section 3.2. Alternatively, the distance travelled can be calculated from trip duration and flight speed (Pearson, 1968) or from provisioning energetics (Flint, 1991), although these tend to overestimate range (Birdlife International, 2000). Foraging ranges vary enormously among species (Birdlife International, 2000), and so need to be applied generically to each important colony at which a species occurs (Table 3.2). However, foraging ranges can also vary among sites (Hamer *et al.,* 2000) and years (Monaghan *et al*., 1994) and this may result in generic boundaries being inappropriate. The main problem with this approach is that seabirds generally use a small proportion of their potential foraging range, and so large areas that are seldom used by birds will be included in the IBA or SPA when radii approaches are employed (Birdlife International, 2000). This problem is most acute for species with large foraging ranges such as petrels and gannets, for which radius based methods are wholly inappropriate (Birdlife International, 2000). However, for those with short foraging ranges and low foraging habitat specificity such as terns, this approach is worthy of consideration, as shown in Figure 3.1 (SEO/BirdLife, 2005, 2006).

**Table 3.2. Recommended limits for foraging radii of seabird species breeding in the ICES area as given by SPEA-SEO/BirdLife (2005).**

| RADIUS AROUND COLONY (TO INCLUDE FORAGING AND/OR MAINTENANCE ACTIVITIES) | | | |
|---|---|---|---|
| **5 KM** | **15 KM** | **40 KM** | **STILL UNKNOWN** |
| *Stercorarius parasiticus* | *Calonectris diomedea* (rafts) | *Stercorarius skua* | *Oceanodroma leucorhoa* |
| *Larus genei* | *Puffinus puffinus* (rafts) | *Larus argentatus* | *Oceanodroma castro* |
| *Sterna albifrons* | *Puffinus mauretanicus* (rafts) | *Larus fuscus* | *Pelagodroma marina* |
| *Sterna nilotica* | *Phalacrocorax carbo* | *Larus marinus* | *Hydrobates pelagicus* |
| *Cepphus grylle* | *Phalacrocorax aristotelis* | *Larus audouinii* | *Fulmarus glacialis* |
| | *Larus ridibundus* | *Rissa tridactyla* | *Morus bassanus* |
| | *Larus canus* | *Uria aalge* | |
| | *Larus melanocephalus* | *Alca torda* | |
| | *Sterna hirundo* | *Fratercula arctica* | |
| | *Sterna paradisea* | | |
| | *Sterna dougallii* | | |
| | *Sterna sandvicensis* | | |

As part of a LIFE Project aimed at identifying marine IBAs in Spain (SEO/BirdLife 2005, 2006), systematic surveys were conducted off the Ebro Delta during the breeding seasons of 2005 and 2006 (Figure 3.1). These revealed that the local breeding tern species (predominantly sandwich and common terns, but also little and gull-billed terns), and most of the breeding gulls (Audouin's and slender-billed gulls) concentrate their foraging activity within the first 5 nautical miles (nm) around the colonies (more than 60% of birds counted), with a further 15% of activity occurring out to 10 nm. These ranges are even narrower if the less coastal Audouin's gull is excluded; in that case 75% of activity concentrates within 5 nm and more than 90% within 10 nm.

These results will help to define marine IBAs based on radii around colonies that will include the main foraging areas for these species.
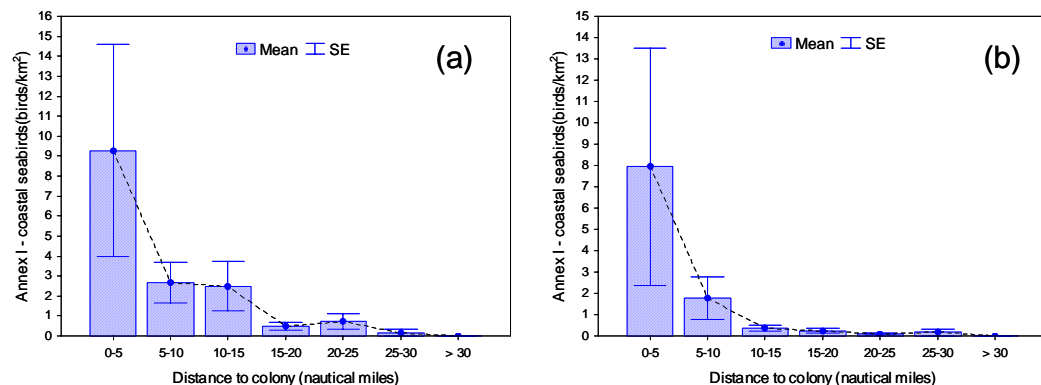


**Figure 3.1. Density (mean ± SE) of breeding gulls and terns included in Annex I of EU Birds Directive with respect to the distance from their colonies at the Ebro Delta, NE Spain. In (a) all local breeding species are included; in (b) Audouin's gull is excluded, on the basis of its less coastal habits, showing a smoother decreasing pattern of density with respect to distance from the nearest colony.**

The following hierarchy of approaches is suggested to maximise the application of available data in order to apply appropriate radii:

1 ) Species x site-specific data (either gathered from literature, or through current field based projects, bearing in mind potential density-dependence due to differences in colony size and other ecological considerations that may determine the size of the

radii. In cases where multi-species colonies exist, the species with the largest foraging radius should be used to set the outer radius).

2 ) Species-specific data.

3 ) If data are not available to apply 1 or 2, then nearest neighbour or surrogate species data may be used.

Specific foraging areas around individual colonies can be identified from empirical surveys of distribution based on either at-sea surveys or tracking studies. These have the potential to identify discrete foraging patches used by seabirds around colonies, and they need not be contiguous with the colony itself. The drawback is that the locations of small discrete foraging distributions may change over years (Monaghan *et al.*, 1994), which may result in inappropriate areas being identified as important. Boundaries need to be sufficiently broad to encompass such changes in foraging distributions and, where these are spatially variable; a radius-based approach may be preferable. Furthermore, identifying specific foraging areas empirically requires data for every species at every colony SPA or IBA and this will usually demand expensive survey work over several years. Alternatively, models of foraging energetics and habitat characteristics can be used to predict colony-specific foraging areas (Wanless *et al.*, 1997) and effects of environmental perturbation on distribution (Wanless *et al.*, 2003). This approach has been applied with some success to predict foraging distribution of European Shag (Wanless *et al.*, 1997, 2003 and is now being developed for common guillemot (Wanless pers. comm.). Such models have potential to provide a generic method to determine the location and extent of colony-specific foraging areas and their annual variability. However, it is advisable to test the predictions of the models using empirical data at several colonies prior to wider application.

**Box 3–1. Determination of possible SPAs for Manx shearwaters using radio telemetry.**

Radio transmitters were attached to breeding Manx shearwaters at three existing terrestrial breeding colony SPAs in the UK – Bardsey, west Wales, Skomer, south-west Wales, and Rum, west Scotland. These birds form dense flocks ("rafts") on the sea before dusk before entering the breeding colony.

In order to identify likely boundaries for an extension of the existing SPAs into the marine environment, the results from radio-tracking birds in rafts were analysed using kernel analysis. This method aims to define the home range or area of greatest use of animals (Powell, 2000).
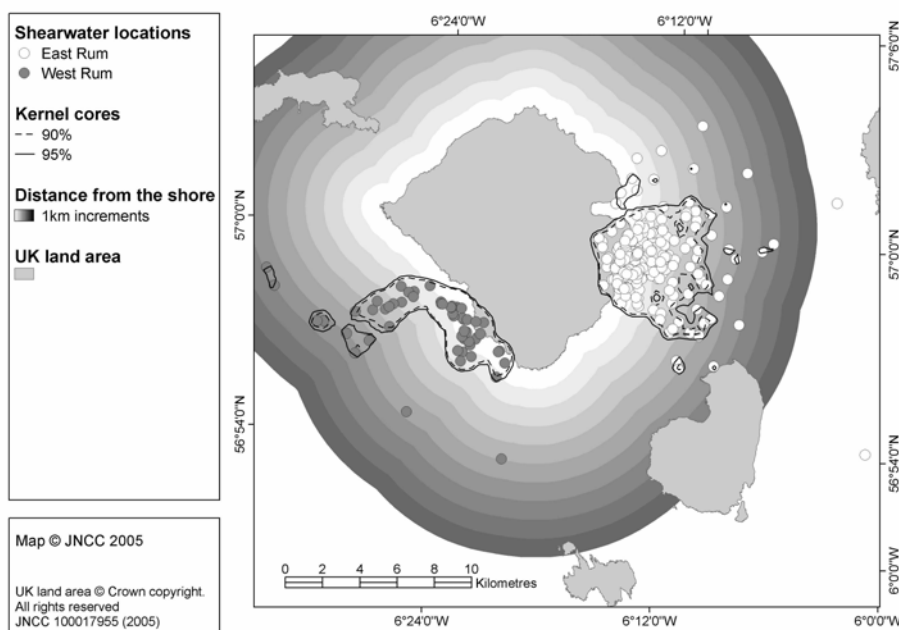


Recommendations for seaward extensions to the SPA boundaries at these three colonies have been made based on the areas within which the birds appear to spend 95% of their time (McSorley *et al.*, 2005). Although an arbitrary proportion 95% seemed a sensible one for three reasons:

a )   there was little difference between the 90% and 95% kernels at all three colonies;

b )   95% is a useful analogy with statistical significance (though it should not be confused with that);

c )   it accords with other studies.

### 3.5.2   Inshore aggregations

Protected areas for inshore concentrations of (mainly) seaduck and divers in the non-breeding seasons have generally been identified using data from aerial surveys. Having determined that an area qualifies to be accorded protection (with respect to numerical thresholds such as population size or density) then the boundary of the site may be determined by a variety of methods. The methods should as far as possible be "objective" and repeatable. They could be applied at various scales from the necessarily very coarse scale if raw survey data only are used, or to the finer scale enabled by interpolative or other spatial modelling of the data.

Whichever method is applied to define boundaries it will inevitable involve one or more subjective judgements such as how much of the distribution of the birds to include within a boundary, whether satellite aggregations disjunctive from any core concentration should be included, which parts of the habitat should be included if using habitat variables to define the boundary. Such judgements should as far as possible be founded on the best available scientific knowledge; political or socioeconomic considerations should not influence them. They should be strictly operational and again as far as possible widely and generically applicable. For example, if a judgement is to be made on how much of a modelled distribution to include in a protected area boundary then sound numerical judgements based by analogy on existing, albeit arbitrary, thresholds might be more be more defensible than seemingly capricious ones. In this context proportions such as 1% and 5% have a resonance, but if applied, they defer to some ecological sense.

### 3.5.3 Offshore concentrations

While there may be data available on offshore distributions of seabirds they are unlikely to be sufficiently extensive both spatially and temporally to allow anything but a very crude assessment of their use in defining protected areas. Usually, some sort of modelling approach will be necessary to define these. This may be either spatial modelling using interpolation based solely on the spatial structure of the data or it may be one based on extrapolation using known seabird-habitat associations.

The 1% criterion is used as a numerically qualifying threshold for both SPA and IBAs, and has gained wide acceptance, again despite its somewhat arbitrary nature and lack of a strict biological basis. Of course, if a large enough area is considered then 1% of the relevant (national/biogeographical) population will certainly be accommodated within the area. The maximum size of an area considered to be internationally important has not been defined. A larger site will contain more birds than a smaller site centred around the same location. The aim of SPAs and IBAs is to protect discrete concentrations of birds rather than extensive areas in which densities are low, so adjustments to the 1% threshold based on the area the concentration occupies may be necessary to avoid the latter occurring.

In order to compensate for area size, Skov *et al.* (2000, 2007) developed the Marine Classification Criterion (MCC) in interpolative spatial modelling. The MCC combines the proportion of the total population and the degree of concentration of the birds to be protected within the area. The proportion of the total population is defined as the estimated number of birds within an aggregation divided by the total population and multiplied by 100. The MCC is achieved if not only 1% of the biogeographical population of a particular species is concentrated in an area (site), but if in addition, the average density at that site exceeds four times the average density in the relevant regional sea. The choice of this critical size of the reference basin-wide density level (four times the average density) was determined both by the requirement to secure the inclusion of those areas host to globally important aggregation and by the need to exclude peripheral areas characterised by moderate densities dispersed over wide areas (Skov *et al.*, 2007).

The MCC allows applying the 1% criterion for the identification of concentrations of seabirds of international importance (see also Skov *et al.*, 2000) while minimising the risk of selecting unduly large areas of sea. Formerly, Skov *et al.* (2000) used an area of 3000 km² as the reference to control for area size instead of the current density comparison. In spite of the fact that the different Marine Classification Criteria (MCC) uses different areas adjustment factors, the differences in the resulting selection of areas were minor. Most areas selected using the early version of the MCC were retained during the most recent analysis, and the boundaries were modified only slightly as an effect of adding or deleting the boundary of a few concentrations of moderate importance (Skov *et a.*, 2007).

An alternative modelling approach might be to apply techniques that model the association between seabirds and their habitats or other physical features of the marine environment. The spatial distribution of seabirds feeding on pelagic prey is less predictive than the distribution of benthos-feeding species simply because the prey of pelagic feeding species fluctuates more in space and time. The spatial position of a concentration of birds might well vary considerably over time as a result of, for example, changes in the position of an upwelling zone or of other hydrographic features that are not fixed in space. The designation of offshore SPA's is made more difficult by this fact. In the open sea there are no visible natural boundaries with which to delineate protected areas. As the distributions of seabird species are determined by hydrographic features then extensive information, if it exists, on such features can be used to optimize SPA site selection. Many marine areas are now covered by hydrographic model platforms. These models store standard hydrographic variables (salinity, temperature, density, current speed and direction) in a four dimensional form (longitude, latitude, depth and time). These hydrographic models vary in their temporal extent. So they can be used to describe the spatial and temporal stability of key features determining bird distributions, inter- as well as intra-annually.

### 3.5.4 Migration hotspots

To date, little development has been achieved in how boundaries of migration corridors or other hotspots might be identified. Potential approaches should follow the same principles of using the best available data and deploying objective and repeatable methods. Where judgements are necessary the same general principles might apply as in determining boundaries of other types of protected area for marine birds.

### 3.5.5 Use of habitat features to determine boundaries

Marine systems may outwardly appear homogenous, but do contain various habitat features that birds associate with at elevated densities (e.g. Hunt and Schneider, 1987). Where birds associate strongly with a habitat feature, this feature may be used in defining the boundaries of an SPA or IBA in similar ways as in terrestrial, freshwater or intertidal systems.

Habitat attributes that may be important for boundary determination include physical or biological features such as bathymetry, temperature, salinity, and chlorophyll. These often result in phenomena such as upwelling zones and frontal areas, where food resources may be rendered more abundant. Such features are sometimes rather short-lived and small (e.g. river plume fronts), but the location and timing of some are quite predictable (e.g. upwelling, shelf break fronts).

Marine habitat features have already been used in the designation of protected areas worldwide (Hyrenbach *et al.,* 2000), and also for SACs within Europe. These have potential to complement seabird conservation where important numbers occur within their boundaries. For example, the SAC Dutch Coastal Sea designated to protect shallow sandbanks encompasses important concentrations of seaduck (Lindeboom *et al.,* 2005; Section 3.5.3), and that for the Friesian front post-breeding concentrations of guillemots (Leopold *et al.,* in press; Section 3.5.3). It has also been recently proposed to use hydrographical clues to identify MPAs protecting the breeding foraging grounds of the Balearic Shearwater (Louzao *et al.,* 2006; Section 3.5.6). For other pelagic species (shearwaters, storm-petrels) current work suggest there are no clear oceanographic variables that may correlate to seabird areas over the years.

MPAs centred on habitat features are often surrounded by buffer zones, which are intended to allow for dispersal of animals associated with it or, in the case of hydrographic features, uncertainty in the location of the feature itself (Hyrenbach *et al.,* 2000). SPAs and IBAs do not include provision for buffer areas around sites.

### 3.5.6 Delimiting boundaries of bird concentrations from transect counts

Transect data are usually interpolated using kriging (Cressie, 1991), a statistical method that produces estimates of density across a fine-scale grid based on counts in adjacent squares and patterns of spatial autocorrelation. This generates a map of bird density over the entire study area. Boundaries around concentrations identified by transect surveys are generally determined by generating density contours (isolines) in GIS. In some studies (Skov *et al.*, 1995, 2000), arbitrary contour intervals were chosen, which means that boundaries are to a certain extent determined by these predetermined limits as well as density. A more quantitative approach is to estimate gradient of bird density change over space. This approach identifies the strongest gradient in spatial density and positions the isoline just outside this, which is then treated as the border of the concentration. In this way, the major part of the concentration is included in the selected area (Garthe, 2006; Garthe and Skov, 2006). In other studies, an important concentration of birds may already be recognised and the challenge is to identify the boundary of this. In these cases, a contour that encloses a given percentage of the aggregation needs to be generated, as for Common Scoters in Camarthen Bay (Webb *et al.*, 2004a; Box 3–2).

**Box 3–2. Determining the boundary of Camarthen Bay SPA.**

Carmarthen Bay, south Wales, hosts internationally important concentrations of Common scoter *Melanitta nigra* outside the breeding season.

Aerial surveys, deploying a standardised methodology (Kahlert *et al*., 2000) were undertaken over the area in winter 2001/2002. Sampled densities of scoter were converted into total population size using distance methodology (Buckland *et al*., 2001).
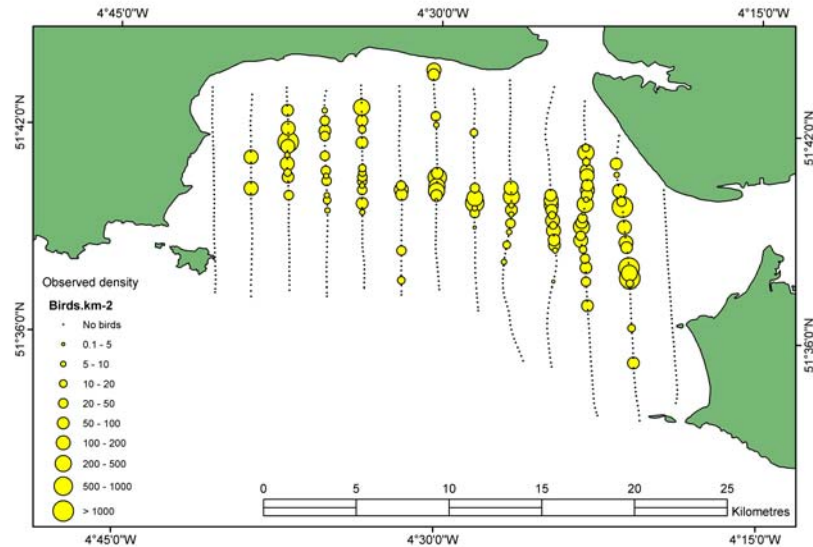


Scoter density was modelled throughout the survey area using kriging, a spatial interpolation method based on variography (Cressie, 1991).
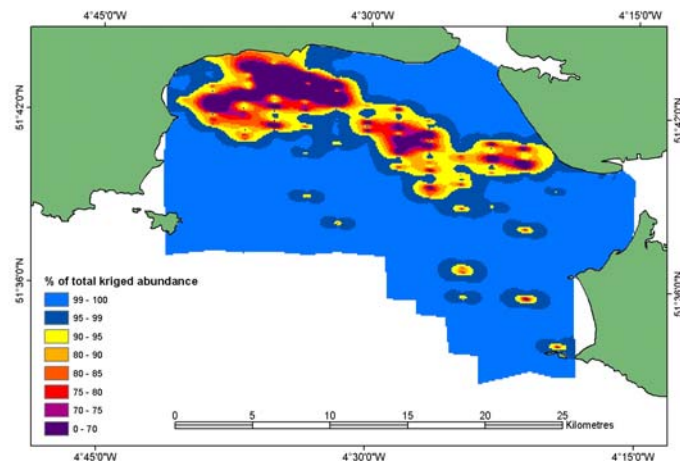


Recommendations for the seaward boundary of the SPA were made, such that 95% of the modelled population was contained within the boundary. See Webb *et al*. (2004a) for full details of this case study. See also McSorley *et al*. (2004) and Webb *et al*. (2004b,c) for further details of the methods applied in identifying sites and boundaries for inshore concentrations of waterbirds outside the breeding season, including rules for the inclusion of satellite aggregations disjunct from the core aggregation.

### 3.5.7 Determining boundaries of bird concentrations from tracking data

For any bird or number of birds that have been instrumented, kernel density estimation provides a means of quantifying habitat use (Georges, 1997; Wood *et al.,* 2000). To calculate kernels, data are first standardized by resampling the tracks followed by each bird at hourly intervals. This process gives the same density of locations per unit time. The resampled tracks are then summed to provide estimates of core areas used by the bird or birds. A smoothing parameter ("h") has to be chosen that represents the spatial scale over which observations are averaged. This smoothing parameter represents the distance between each location and the nearest grid intersection on the surface to which the birds' core range is to be plotted. The value of h chosen can significantly affect the results (Hemson *et al.*, 2005), especially for animals that often return to a home point such as a nest or roost. This statistical problem can be minimized if a relatively large number of birds are tracked (BirdLife International, 2004). Concentric polygons are then drawn around areas encompassing some percentage of the total area visited by all birds (BirdLife International, 2004).

Kernel density estimates can be weighted by extrapolating to the total population within which individuals were selected for tracking. The weighted density estimates can then be plotted.

For some species such as petrels, which forage very far from their nesting locations, kernel density estimation could give misleading results, in that the area close to the breeding colony will emerge as important even if no feeding is done in its vicinity. Data from areas very far from their breeding grounds will also be a small percentage of their total use, therefore only allowing to continue boundary-identification through environmental-data correlation. Because of this, it may be advisable to remove commuting data points in order to highlight important foraging habitats. For migrating birds tracked by GLS, kernel estimators are not appropriate. Instead, one calculates mean latitude ± 1 SD within each 10 degree band of longitude and plots the results (BirdLife International, 2004).

## 3.6 National approaches

### 3.6.1 Denmark

The Danish Forest and Nature Agency (Ministry of the Environment) is the administrative body responsible for classifying Danish SPAs; details can be found at http://www.skovognatur.dk/English/.

The following guidelines are applied in selecting areas for possible SPA classification in Denmark:

- Areas holding annex I species that breed regularly in the area in numbers of 1% or more of the national population.
- Areas holding abundant annex I species that appear in numbers of international importance during part of their life cycle or areas holding scarcer annex I species in numbers of more than 1% of the national population.
- Areas holding restricted but important numbers of widely distributed species
- Areas with regular appearance of migratory species in numbers of 1% or more of the flyway population.
- Areas with regular appearance of more than 20.000 waterbirds (excluding gulls).
- Areas holding restricted but important populations that significantly support the presence within the whole distribution area of a species in Denmark.
- Areas holding restricted but important populations, significant for the survival of a species during critical periods of their life cycle.

In marine areas, designation has been effected primarily with reference to Article 4.2 of the EU Birds Directive; i.e. migratory species that occur in internationally important concentrations.

In Denmark, an initial 111 areas were designated as SPAs in 1983, with preliminary boundary definitions. In 1994, precise delineation of these sites was finalised, the 111 SPA sites covering an area of 976 000 Ha. Of these, 47 SPAs were intertidal/near-shore areas or over shallow marine water. These constitute around 60% of the entire area covered by SPAs. Figure 3.2 indicates the location and extent of SPAs in Danish waters.
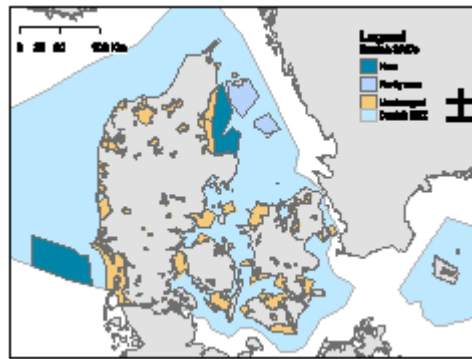


**Figure 3.2. The geographical distribution of Danish SPA's. In 2002 and 2003 two new SPA's were designated, while another two marine SPA's were enlarged.**

Site selection of coastal SPAs was made using available data. In terrestrial and intertidal areas, data were provided by the Danish Ornithological Society whereas those for marine areas were provided by a long-term monitoring programme on waterbirds in inner Danish waters.

In 2002, the process of adding more marine areas to the SPA suite of sites was initiated, in response to the EU Commission. This led to the classification of another two SPAs in 2003. Of these, one was designated primarily for species foraging on benthos, while the other was entirely offshore and designated for red-throated diver and little gull. This added another 530 000 Ha to the total area of marine SPAs. In addition, two existing SPAs were enlarged in 2002, the aim being to improve inclusion of the interest species, common scoter and common eider.

A national monitoring programme has been established with the aim of assessing whether those species for which SPAs have been classified are of favourable status. As part of this programme, waterbirds are monitored every three winters (January/February) in all inner Danish waters, providing estimates of total numbers and distribution. Similarly, moulting diving ducks are monitored in selected areas every sixth summer (August).

### 3.6.2  Germany

In Germany, responsibility for designating marine protected areas is split between different administrations. Generally, nature conservation is the responsibility of the Federal States. As national territorial limits extend out to only 12 nm, responsibility for the German parts of the North Sea is now divided among the Federal State of Lower Saxony (within the 12 nm zone, southern part), the Federal State of Schleswig-Holstein (within the 12 nm zone, northern part), and the Central Government (for the EEZ). In the German Baltic, there are responsibilities allocated to the Federal State of Schleswig-Holstein (within the 12 nm zone, western part), to the Federal State of Mecklenburg-Vorpommern (within the 12 nm zone, eastern part), and the Central Government (for the EEZ). This complicates the procedure of selecting suitable areas because the process is not well coordinated between the different governmental units, and also

because these regions are not easily manageable (e.g. the EEZ in the Baltic is merely a small strip in some German areas).

In order to meet legislative requirements, the Federal States began designating SPAs under the Birds Directive a few years ago, the first area being announced in October 1997. Progress with SPA classification developed differently in the German EEZ, which was declared in November 1994. As this area is not German territory, designation depended for a long time on resolution of international rules/laws/conventions because it was unclear whether SPAs could be designated outside national territorial limits. Furthermore, it was not until April 2002 that it was possible to designate SPAs within the EEZ because of national legislation. At that time, the national law for Nature Conservation was changed and the responsibilities for the designation of marine protected areas were clarified. The Federal Agency for Nature Conservation is responsible for selecting potential protected areas, whereas the Federal Environmental Ministry is responsible for designation/submission to the European Commission.

Under politically induced time-pressure, proposals for SPAs in the EEZ were not addressed until 2002 (see Box 3–3; Garthe, 2003, 2006; and Garthe and Skov, 2006). Two areas were proposed and designated in May 2004 with slightly altered geographical borders. The Federal State of Schleswig-Holstein finally designated all SPAs in 2005; the Federal States of Niedersachsen and Mecklenburg-Vorpommern are still in the process of designating the remaining sites.

**Box 3–3. Identifying marine SPAs in the German EEZ of the Baltic Sea.**

Using standardised transect survey methods (Webb and Durinck, 1992) data on the distribution of all birds at sea were collected between 1987 and 2002; more than 15,000 ship km were travelled. Distance sampling analyses (Buckland *et al.*, 2001) were applied to the data for all EU Birds Directive Annex I (n=7) and migratory (n=13) species in order to estimate total numbers of birds present. Ordinary kriging (Cressie 1991), an interpolation technique that uses the spatial autocorrelation in the raw dispersion data, enabled the modelling of continuous dispersion throughout the survey area. The modelled distributional data were projected onto a two-dimensional map for each species. For example, long-tailed duck *Clangula hyemalis*:
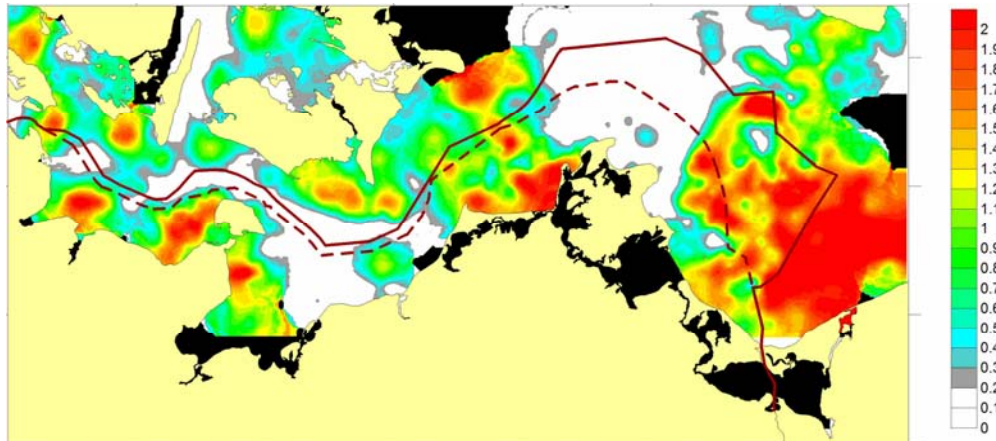


Individual species maps were combined to depict areas of overall importance. Boundaries between high concentration areas were determined by analysing the gradient of modelled bird density change over space, thereby allowing the identification of potential SPAs.

Based on this procedure, a single large SPA of c. 2,000 km² in the German EEZ of the Baltic Sea has been classified (red shading). The dashed line indicates the limit of German territorial waters, the continuous line the limit of the German EEZ. This EEZ SPA is defined by overlapping concentrations of several species, primarily by the distribution and abundance of Slavonian grebe, long-tailed duck, common scoter, velvet scoter and black guillemot, and additionally red-throated diver, black-throated diver and red-necked grebe. This SPA complements those identified in inshore waters of the German Baltic Sea (blue shading).
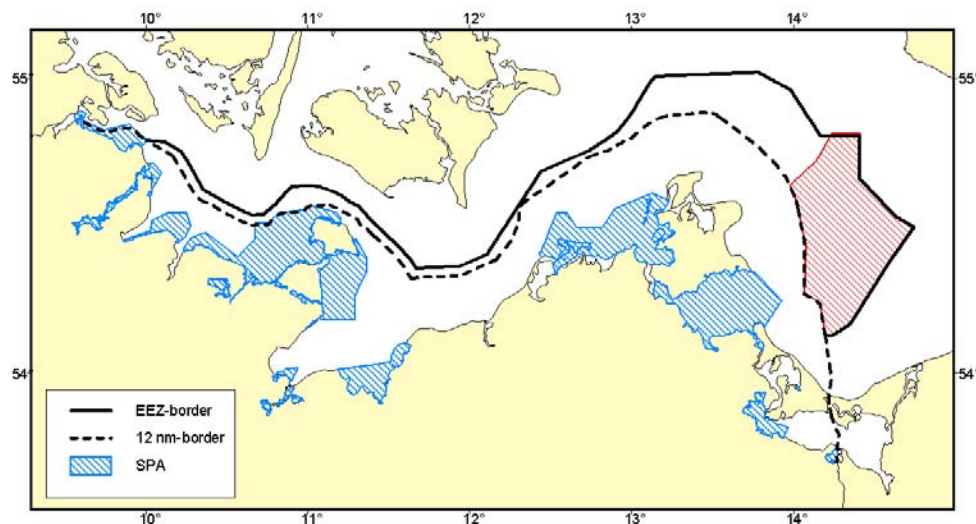
### 3.6.3 The Netherlands

The Netherlands has designated two special protection zones under both the EU Birds Directive and the EU Habitats Directive. Four more "Areas with specific ecological values" are currently under review and might soon achieve similar status; another three are also being reviewed but will require further study (Lindeboom *et al.*, 2005). The two designated areas ("Voordelta" and "Wadden Coast") are special protection zones under both the EU Birds Directive and the EU Habitats Directive. These areas are in fact the southern and northern parts of a continuous strip of shallow (0–20 m deep) water, running along the entire length of the country. It is recognized that the mid-section of this area also potentially holds important bird numbers, but rather than designating all coastal waters, the Netherlands has opted for protection of the richest parts. Key bird species are red-throated diver, great crested grebe, common scoter, common eider, and all *Larus* and *Sterna* species breeding along the Dutch coast, migrating along the coast, or wintering in the nearshore waters (gulls only). The Netherlands has decided, in the "National Spatial Strategy" that more SPAs should be identified, particularly in the offshore parts of the EEZ, taking into consideration requirements both of the Birds and the Habitats Directives and the OSPAR Convention. Seabirds are therefore part of the equation, but other ecological values also play a significant role, both in identifying these areas and in defining their boundaries. Three such areas have been determined: "Dogger Bank", "Central Oyster Grounds", "Cleaver Bank" and "Frisian Front" (see Figure 3.3). The first two of these have no specific bird interest, but the Cleaver Bank was found to have a higher than average diversity of seabirds, and the Frisian Front was selected because of regularly occurring vulnerable concentrations of common guillemots with chicks in summer (Leopold *et al.*, in press). The boundaries of these areas however, are largely determined by physical and benthic features, although in the case of the Frisian Front it was confirmed that the core area for the guillemots was well within these boundaries. Areas that will be considered in the future include an area with natural gas seeps (with little if any relevance for seabirds), a nearshore "reef" area at the Dutch/German border (mainly of interest because of the benthos, but possibly also important for divers), an area in the central Southern Bight of the North Sea, where large numbers of auks occur in late winter, and an area in the south-west with extensive shallow banks (which also might have high bird interest). In summary, while at least three areas have been specifically (co-)selected for seabirds, most proposed or studied areas have been or will be selected for more than one purpose, fulfilling Birds and Habitats Directives requirements, as well as OSPAR criteria.
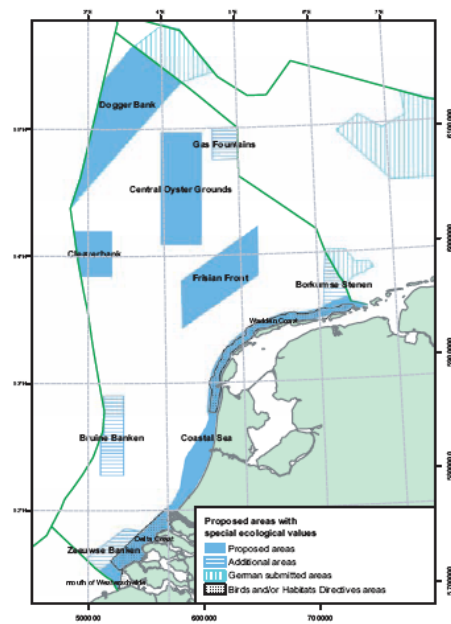
**Figure 3.3. Map of the Dutch EEZ with all (potential) "areas of specific ecological values". Solid blue areas are those that have either already been designated (Delta Coast and Wadden Coast) or have recently been proposed by Lindeboom *et al.*, 2005). Hatched areas will be considered after further study.**

### 3.6.4    Belgium

The general process followed in Belgium for classification of SPAs contrasts with that in other Member States. SPAs are classified for Annex I and migratory species only if those species are accorded protection under other international conservation instruments.

In 2001, a Royal Decree was issued in Belgium legally protecting all birds in Belgian marine waters. Belgium has not yet designated any SPAs in marine waters, but Haelters *et al.* (2004) has outlined the process for marine SPA classification in Belgium. This follows a formal assessment process. Those species of waterbird on Annex I of the EU Birds Directive that occur regularly in qualifying numbers in Belgian waters will have SPAs considered for their protection. However, only those regularly occurring migratory species not on Annex I that are already protected by another international convention(s) will potentially be accorded protection within SPAs. Hence, unless such species are already protected by some other instrument no SPAs will be classified for them. Haelters *et al*. (2004) propose three SPAs targeting the protection of seven species (three Annex I species and four migratory species). These areas were assessed firstly on the concentration of birds they host, resulting in the identification of the most suitable habitats for great skua and little tern. For the other five species (Sandwich tern, common tern, great crested grebe, common scoter, and little gull) the most suitable habitats in number and size were selected for each species and the areas overlaid to assess the final area for possible SPA status. From this analysis, three areas were identified, two of which extend from low water mark out to 6 nm offshore (off Koksidje and De Panne and off the coast from Middeelkerke to Bredene); the other area is focused around the harbour of Zeebrugge).

### 3.6.5    United Kingdom

Guidelines to determine terrestrial SPA qualification in the UK are well-established (Stroud *et al.,* 2001). As far as has been possible these guidelines have been applied in the marine environment also. SPA classification process is a two stage process. In the first stage, an area is deemed to qualify as an SPA if:

- it is used regularly by 1% or more of the Great Britain population of a species listed in Annex I of the Birds Directive in any season;

- it is used regularly by 1% or more of the biogeographical population of a regularly occurring migratory species (other than those listed in Annex I of the Birds Directive) in any season;

- it is used regularly by more than 20 000 waterfowl (as defined by the Ramsar Convention) or 20 000 seabirds in any season.

Stage 2 of the process further considers candidate areas on the strength of their importance to species with regard to population and ecological attributes, such as rarity of the species concerned, naturalness, population density, range characteristics, and others. If a site qualifies as an SPA it may not necessarily be taken forward to classification. The overall aim should be to establish a network of sites that is coherent.

Considerable progress has been made in the UK over the past few years in identifying marine SPAs. The general approach has been one of identifying important sites in three broad categories – a) marine extensions to existing breeding seabird colony SPAs; b) sites for inshore concentrations of waterbirds in the non-breeding seasons; and 3) sites for offshore aggregations of seabirds. To date, recommendations have been made and endorsed to extend existing SPAs for common guillemot, razorbill, and Atlantic puffin by 1 km into the marine environment, by 2 km for northern fulmar and northern gannet breeding SPAs, and by at least 4 km for Manx shearwater SPAs (see Box 3–1). Recommendations have also been made not to extend, at least until further planned work has been completed, those SPAs for which storm-petrels, European shag, and terns are interest features; and recommendations have been made not to extend existing breeding site SPAs for which great cormorant, skuas, gulls, black-throated diver, great crested grebe, Slavonian grebe, common scoter, or red-necked phalarope are interest features.

Possible SPAs have been identified in two inshore sites, one for red-throated diver and one for both common scoter and red-throated diver, and one wholly marine SPA has been classified for common scoter (see Box 3–3). Analyses have been completed for a further two areas that host qualifying numbers of divers. An aerial survey programme covering all the important inshore areas around the UK continues. Plans have been drafted to explore the possibility of identifying important concentrations of Balearic shearwater in the Channel, and of identifying possible SPAs for offshore aggregations of seabirds. The latter aims to analyse the European Seabirds at Sea database with a view to determine possible SPAs in the breeding and non-breeding seasons. Spatial modelling (interpolation) will result in density surfaces for several species to which apply the Marine Classification Criterion might be applied.

### 3.6.6   Spain

In Spain, the recognition of SPAs in the marine environment by the Birds Directive is currently limited to only 20 coastal sites (574 km$^2$)[1]. On the other hand, under the Habitats Directive, there are 88 Sites of Community Interest (SCI) have been already designated (5191 km$^2$)[2], basically corresponding to inshore and coastal areas. However, The Spanish Ministry of Environment (MMA) is currently preparing a new inventory of SCIs, with particular focus in offshore areas.

Spain comprises three biogeographical marine regions: Mediterranean, Atlantic and Macaronesia. These regions host a wide diversity of seabird species, many of them of conservation concern, and which face various human pressures (Madroño *et al.*, 2004). Among these pressures, more or less effective protection exists on the land, particularly at

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1 http://ec.europa.eu/environment/nature/nature_conservation/useful_info/barometer/pdf/spa.pdf

2 http://ec.europa.eu/environment/nature/nature_conservation/useful_info/barometer/pdf/sci.pdf

colonies, but little effort has been addressed in the open sea, where threats are not well understood but seem to pose a serious threat for some populations (Madroño *et al.,* 2004).

The Mediterranean region holds significant populations of species of high conservation concern, which are subject to strong pressure by human activities such as fisheries (reviewed in Arcos *et al.*, in press). Seabirds of interest include the whole breeding population of the critically endangered Balearic shearwater and about 90% of the breeding population of the threatened Audouin's gull. The Macaronesian region (Canary Islands) holds significant populations of highly pelagic species, such as Bulwer's petrel*,* little shearwater and Cory's shearwater. The Atlantic region, although hosting only a few relevant breeding species, is a highly productive area of importance for migrating and wintering seabirds.

As a first step in the designation of marine SPAs, particularly regarding offshore areas, the Spanish Ornithological Society (SEO/BirdLife) is conducting a LIFE-Nature project to develop criteria for and identify IBAs at sea. The ultimate target of this project is to obtain a complete and detailed inventory, applying objective methodological criteria, of Important Bird Areas at sea for the seabird species listed in Annex I of the Bird Directive with populations in Spain. Other seabird species will also be considered, but are not explicitly targeted by this project. It is expected to develop, along with a Portuguese sister project run by SPEA – see Section 3.6.7), a standard methodology for the identification and delimitation of IBAs at sea, which could be applicable to other countries.

The existing data on seabird distribution at sea for Spain will be considered, but most data are expected to come directly from transect surveys conducted within the framework of the project, as well as the tracking of breeding seabirds, mainly using satellite telemetry of Cory's shearwater and Audouin's gull. Habitat features will be used to model seabird distribution patterns, and the use of these to define IBAs boundaries will be considered (see Louzao *et al.*, 2006; Box 3–4).

**Box 3–4. Understanding the oceanic habitat of the critically endangered Balearic shearwater to identify suitable areas for protection.**

Data on Balearic Shearwaters distribution at sea were collected throughout the Iberian Mediterranean coast, using vessel-based transect survey methods, during the chick-rearing period of 1999-2000 and 2002 (Louzao *et al.*, 2006). The overall foraging range was first identified using presence/absence data, and corresponded to the frontal systems along the eastern Iberian continental shelf (<200 m depth). Main foraging grounds within that area, identified by high density aggregations, were characterised by elevated chlorophyll a concentration.
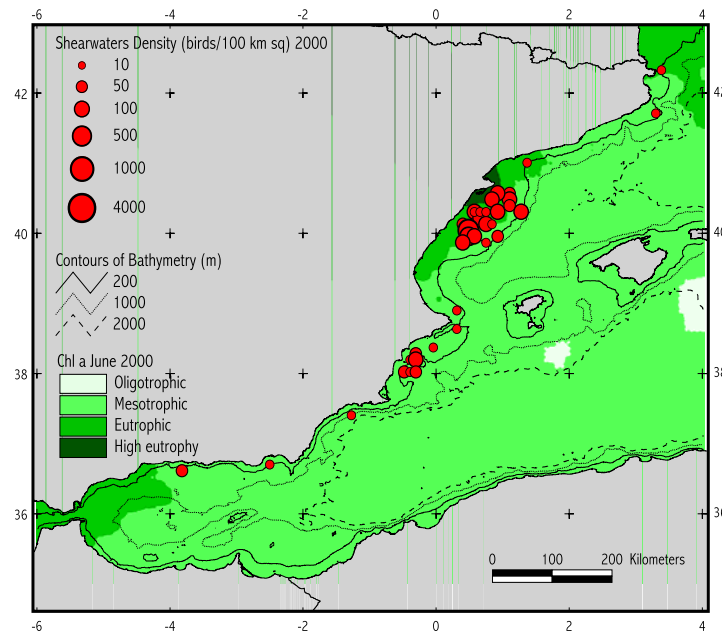


It was proposed to establish a core-buffer protection model, in which the main foraging grounds (i.e. the core region - area of influence of the Ebro River discharge and Cape La Nao region) deserved enhanced protection, while in the remaining foraging range (buffer region) more diffuse protection would be applied.

**Box 3–4. Continued.**

A predictive model based mainly on Chl a gradients properly identified the main foraging range for Balearic Shearwaters for each of the 3 years of study. The use of habitat features to assess suitable areas among years and their overlap could help defining IBA/SPA boundaries.
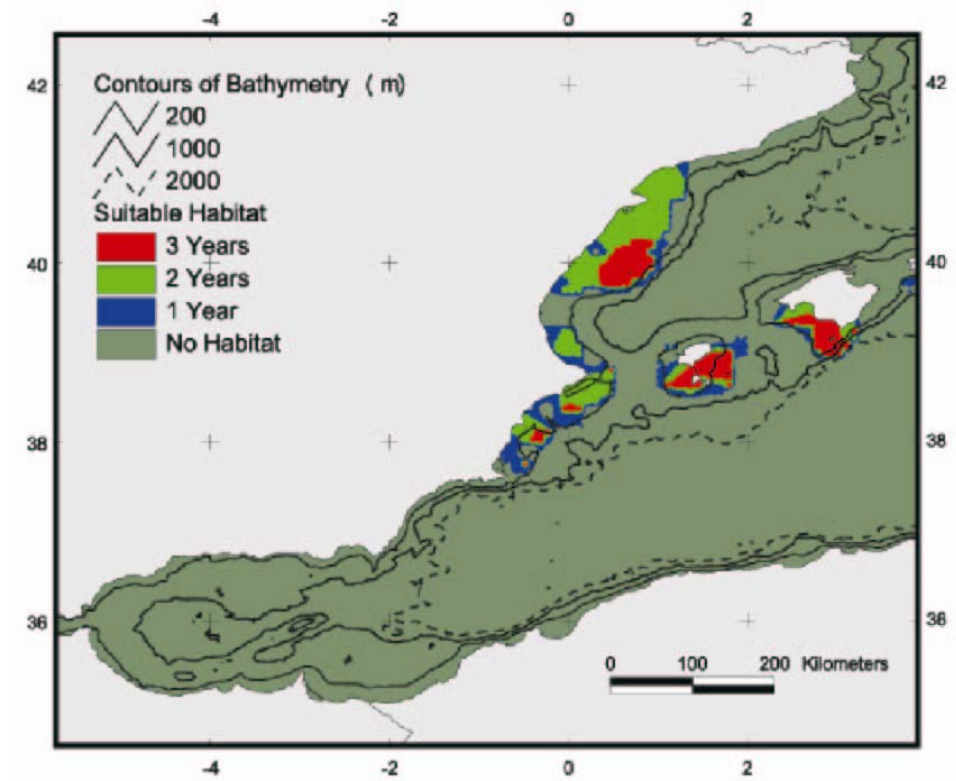
### 3.6.7 Portugal

The oceanic area under Portuguese jurisdiction is eighteen times the size of the land area. The Portugal EEZ totals 1727 408 km², making it the largest in the European Union and the eleventh largest in the world.

In Portugal, the extension of the Habitats and Birds Directives to the marine ecosystem is still at an early stage. The process has been initiated in Azores Archipelago with regards to habitats. The Azores archipelago has already listed offshore locations such as the SACs "Formigas Bank and Dollabarat" and "Dom João de Castro Bank". In the mainland, it was so far classified as SPA a seaward extension of the Berlengas seabird colonies (SPA Berlengas and Farilhoes) and also an inshore area (up to 20 m depth) for wintering scoters (SPA Aveiro). A SAC was proposed to protect the last sea-grass prairie (*Zostera* sp) in the marine environment in Portugal (SAC Arrabida/Espichel). In Madeira Archipelago, SPA designated for the main seabird breeding colonies of Desertas and Selvagens include a seaward extension of 100 m to 200 m depth. This is clearly insufficient to protect most breeding and wintering seabird species occurring in Portugal.

The scales at which seabird dispersion occurs in the Portuguese marine area render it quite difficult to define and characterise seabird densities. The possibility of identifying inshore-coastal aggregations of some wintering species, such as Balearic shearwater and scoters, may lead to the classification of a small number of SPAs off continental Portugal, and recent data from SPEA's project may support this.

Terrestrial seabird colonies that have a coastal component and are already classified might be extended into the sea to protect feeding, resting and/or rafting aggregations of birds. This may include important coastal feeding areas in the breeding season for species such as terns, and species-specific seasonal concentrations, such as "rafting" Manx and Cory's shearwaters during the breeding season. However, such seaward extensions could not integrate all foraging/staging grounds of most threatened seabird species or those Annex I species for which Portugal has highest responsibilities. In this sense, offshore areas hosting concentrations of seabirds are probably Portugal's main challenge for marine SPA designation. Data are currently being collected as truly pelagic species such as *Pterodroma madeira, Pterodroma feae, Pelagodroma marina* or *Oceanodroma castro* require large amounts of data and intense surveys to enable sensible proposals for future SPA classification. Their behaviour at sea is poorly known and the methodologies used to track them are still under development.

Between 2004 and 2008, the Sociedade Portuguesa para o Estudo das Aves (SPEA) is conducting a strategic LIFE-Nature Project to contribute to the implementation of the Birds Directive in the marine environment through a detailed inventory of the most significant marine areas for seabirds included in Annex I of the Birds Directive. This will aim also to formulate adequate methodological criteria for the identification and delimitation of IBAs off Portugal. Some of Europe's rarest bird species will be addressed, such as the globally threatened *Puffinus mauretanicus, Pterodroma madeira and Pterodroma feae*, and many seabird species that are of conservation concern, e.g. *Calonectris diomedea, Puffinus assimilis, Larus audouinii* and *Sterna sandvicensis*.

In 2006, and with a duration of four years, a new LIFE funded Project: "SOS Freira do Bugio" has begun to study seabird presence around the Madeiran arquipelago, with special emphasis on *Pterodroma feae*, therefore allowing better implementation of the IBA/SPA network at sea for this rare and very dispersed seabird species.

In order to achieve this, both projects will link the relationship between oceanographic variables such as temperature, productivity, currents etc. and the occurrence of seabirds will be studied. Monitoring of certain species is being carried out using data-logger, GPS loggers,

and satellite tracking (Cory's shearwater) and radio tracking (Bulwer's petrel, little shearwater, Band-rumped storm-petrel), survey of coastal waters, analysis and mapping of ringing recoveries in Portugal, and the creation of a database of beached birds. The identification and generic sampling of the most favourable areas is being carried out based on the oceanographic characteristics of the areas concerned.

In 2006, radio transmitters have been attached to Bulwer's petrels, band-rumped storm-petrels and roseate terns in order to establish their behaviour at sea, their main movements to and from the colony, and also to gather data to inform a possible seaward extension of an IBA.

The results indicated that the investment needed in terms of time, money and human resources is not justified for Marine IBA classification. Aerial support was not conclusive when tracking these species and locations from air or land where very poor and only of birds flying very close to their colonies. Table 3.3 presents some of the results from this tracking.

**Table 3.3. Information from radio-tracking small Procellaridae in Atlantic waters of Portugal and Spain.**

| SEABIRD SPECIES | LOCATION | NUMBER OF TAGGED BIRDS | NUMBER OF DAYS/HOURS WITH AERIAL SURVEYS | NUMBER OF DAYS WITH CONSTANT TRACKING FROM THE COLONY | NUMBER OF LOCATIONS OBTAINED AWAY FROM THE COLONY |
|---|---|---|---|---|---|
| *Oceanodroma castro* | Alegranza (Canaries) | 31 | 3/12 | 0 | 1 |
| *Bulweriia bulwerii* | Deserta grande (Madeira) | 6 | | 13 | 0 |
| *Oceanodroma castro* | Ilhéu da Praia, Graciosa (Azores) | 4 | | 14 | 1 |
| *Sterna dougalli* | Ilhéu da Praia, Graciosa (Azores) | 4 | | 14 | 2 |
| *Oceanodroma castro* | Berlengas (Continental Portugal) | 20 | 3/15 | 20 | 0 |

### 3.6.8 USA

The official U. S. definition of marine protected area is: "Any area of the marine environment that has been reserved by federal, state, territorial, tribal or local laws or regulations to provide lasting protection to part or all of the natural or cultural resources therein" (Executive Order 13158). The USA initiated a program for the designation of Marine Protected Areas in 2003, and by 2006 there have been between 1500 and 2000 sites designated. Some of these are administered by five federal programs: the National Park Service, the US Fish & Wildlife Service, and three branches of NOAA – National Estuarine Research Reservation program, the National Marine Fisheries Service and the National Marine Sanctuary Program. Other sites are under the jurisdiction of 22 states and territories including Guam, the U.S. Virgin Islands, American Samoa, and the Mariana Islands.

Many of the sites had already been protected under existing regulations, so that listing them as Marine Protected Areas provided no additional protection. For example, the Monomoy National Wildlife Refuge on Cape Cod, Massachusetts, has recently been designated an MPA, but it already had the most restrictive regulations possible in effect due to its status as a Wilderness Area.

Restrictions placed on each site vary according to what species are present and what sort of protection they need. Examples of reserves developed specifically for seabirds include Monomoy Refuge, Massachusetts "Momomoy provides habitat for hundreds of species of resting, feeding and migratory birds. The refuge is so important to migratory shorebirds, in 1999 the Monomoy Islands were designated a Western Hemisphere Shorebird Reserve

Network (WHSRN) regional site. The refuge supports the largest nesting colony of common terns in the Gulf of Maine and the second largest on the Atlantic Seaboard with close to 12000 nesting pairs in 2006." And the Cordell Bank Marine Sanctuary, California "Cordell Bank provides important habitat for many species of groundfish, and is a feeding area for seabirds and marine mammals."

### 3.6.9 Canada

**Protected areas**

Protected marine areas can be created in Canada under legislation administered by three federal agencies: Department of Fisheries and Oceans (DFO), Parks Canada and Environment Canada. DFO has been given the lead in Canada to develop a network of marine protected areas in the Pacific, Arctic and Atlantic Oceans.

DFO legislation specifically mentions the conservation and protection of marine species within Marine Protected Areas (MPAs), but here, the term "marine species" refers only to those administered by the department, i.e. fish, seals, cetaceans, turtles, and invertebrates. Conservation and protection of marine birds can be used only as supporting information in DFO's MPAs, not as the sole reason for their establishment. Currently there are four MPAs in Atlantic Canada, one officially designated, and three announced (2005). None commands particular interest in relation to seabird conservation.

Parks Canada has a plan to establish one or more National Marine Conservation Areas (NMCA) in each ocean region of Canada. There are 13 designated marine regions broadly considered to be Atlantic (10 proper and 3 in high Arctic). To date there has been only one NMCA established in Canada; the Saguenay St. Lawrence Marine Park is located at the confluence of the Saguenay River and the St. Lawrence Estuary. This is an area of high productivity important for cetaceans, seabirds and other marine organisms.

The main mechanism to conserve and protect marine areas important to seabirds in Canada is under legislation administered by Environment Canada. These allow for the establishment of Migratory Bird Sanctuaries (MBSs), National Wildlife Areas (NWAs), which can be comprised of terrestrial and/or marine components. In addition, recent amendments allow for the establishment of Marine Wildlife Areas in the Exclusive Economic Zone (EEZ) outside of territorial waters. Even though the first MWA is still in the planning stages, Environment Canada's existing system of MBSs and NWAs encompass more than 3 million ha of marine habitats making them currently the premier tool for protecting marine habitats in Canada, not just for seabirds, but for all marine species.

The Canadian provinces also have jurisdiction over nearshore waters and there are several examples of protected areas that include a marine component. For example, Newfoundland and Labrador protect five of the most important seabird breeding colonies in that province, under their Wilderness and Ecological Reserves Act. The Act allows for the protection of waters adjacent to the colonies out to a maximum of three nautical miles.

**Identified important marine areas with no formal protection (Atlantic)**

IBAs: Of the approximately 238 IBAs identified to date in provinces and territories bordering the Atlantic (from Nunavut in the north to the Maritimes in the south), 206 (87%) contain marine/open sea component. Almost all are coastal or nearshore and only four contain no land component. One of these is the Lancaster Sound Polynya, which is the only real offshore IBA in Atlantic Canada. The most important seabird colonies are contained within these IBAs and many also include a marine component in the vicinity of the colony.

Others: DFO has recently proposed an initiative to identify Ecologically and Biologically Significant Areas (EBSAs) in the marine environment. Consultation has been broad and has

included consideration of areas important to marine birds. EBSAs can be located anywhere on the continental shelf from the coast out to the 200 mile EEZ.

## 3.7 References

Arcos, J. M., Louzao, M., and Oro, D. *in press*. Fisheries Ecosystem Impacts and Management in the Mediterranean: Seabirds Point of View. *In*: J. Nielsen, J. Dodson, K. Friedland, T. Hamon, N. Hughes, J. Musick, and E. Verspoor (Eds). Proceedings of the Fourth World Fisheries Congress: Reconciling Fisheries with Conservation. American Fisheries Society, Symposium 49, Bethesda, Maryland, in press.

Adams, J., Takekawa, J. Y., and Carter, H. R. 2004. Foraging distance and home range of Cassin's Auklets nesting at two colonies in the California Channel Islands. Condor, 106: 618–637.

BirdLife International. 2000. The development of boundary selection criteria for the extension of breeding seabird Special Protection Areas into the marine environment. Report to OSPAR Commission BDC 00/8/3-E, Vlissingen.

BirdLife International. 2004. Tracking Ocean Wanderers: The global distribution of albatrosses and petrels. Results from the Global Procellariiform Workshop, 1–5 September 2003, Gordon's Bay, South Africa. Cambridge, U. K. BirdLife International.

Birdlife International. 2005. Towards the identification of marine IBAs in the EU: an exploration by the Birds and habitats Directives Task Force. Unpublished report. BirdLife International, Cambridge.

Buckland, S. T., Anderson, D. R., Burnham, K. P., Laake, J. L., Borchers, D. L., and Thomas, L. 2001. Introduction to Distance Sampling, Estimating Abundance of Biological Populations. Oxford University Press 2001, 432 pp.

Camphuysen, C. J. ed. 2005. Understanding marine foodweb processes: an ecosystem approach to sustainable sandeel fisheries in the North Sea. IMPRESS Final Report. NIOZ-rapport 2005–5.

Camphuysen, C. J., Fox, A. D., Leopold, M. F., and Petersen, I. K. 2004. Towards standardised seabirds at sea census techniques in connection with environmental impact assessments for offshore wind farms in the UK. Report for COWRIE, 38 pp.

CEC. 2006. Reasoned Opinion Case C-418/04, Commission of the European Communities v. Ireland 14 September 2006.

Christensen, T. K, Petersen, I. K., and Fox, A. D. 2006. Effects on birds of the Horns Rev 2 offshore wind farm: Environmental Impact Assessment. NERI Report, Commissioned by Energy E2. 79 pp.

Cochran, W. W. 1980. Wildlife telemetry. *In* Wildlife Management Techniques Manual 4 th edition. Ed. by S. D. Schemnitz. pp. 507–520. Wildlife Society, Washington.

Cressie, N. A.C. 1991. Statistics for spatial data. Wiley-Inter-Science, New York.

Davis, L. D., and Miller, G. D. 1992. Satellite tracking of Adélie Penguins. Polar Biology, 12: 503–506.

EEC. 1979. Council Directive 79/409/EEC of 2 April 1979 on the conservation of wild birds. Official Journal L103 (25.4.1979). Official Journal L103 (25.4.1979).

EC. 1992. Council Directive of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora 92/43/EEC.

EU. 2006. Guidelines for the establishment of the Natura 2000 network in the marine environment. Application of the Habitats and Birds Directives.

Flint, E. N. 1991. Time and energy limits to the foraging radius of sooty terns *Sterna fuscata*. Ibis, 133: 43–46.

Freeman, A. N.D., Nicholls, D. G., Wilson K.-J., and Bartle, J. A. 1997. Radio- and satellite-tracking westland petrels *Procellaria westlandica*. Marine Ornithology, 25: 31–36.

Garthe, S. 1997. Influence of hydrography, fishing activity and colony location on summer seabird distribution in the southeastern North Sea. ICES Journal of Marine Science, 54: 566–577.

Garthe, S. 2003. Verteilungsmuster und Bestände von Seevögeln in der Ausschließlichen Wirtschaftszone (AWZ) der deutschen Nord- und Ostsee und Fachvorschläge für EU-Vogelschutzgebiete. Berichte zum Vogelschutz, 40: 15–56.

Garthe, S. 2006. Identification of areas of seabird concentrations in the German North Sea and Baltic Sea using aerial and ship-based surveys. *In* Advancing towards effective marine conservation in Europe: Natura 2000 sites in German offshore waters. Pages 225–238. Ed. by H. von Nordheim, D. Boedeker and J. C. Krause. Springer.

Garthe, S., Montevecchi, W. A., and Davoren, G. K. 2007. Flight destinations and foraging behaviour of northern gannets (*Sula bassana*) preying on a small forage fish in a low-Arctic ecosystem. Deep-Sea Research II 54: 311–320.

Garthe, S., and Skov, H. 2006. Selection of suitable sites for marine protected areas for seabirds: a case study with Special Protection Areas (SPAs) under the EU Birds Directive in the Baltic Sea. *In* Waterbirds around the world. Pages 739–742. Ed. by Boere, G. C., Galbraith, C. A., and D. A. Stroud. The Stationary Office, Edinburgh, UK.

Georges, J. Y., Guinet, C., Jouventin, P., and Weimerskirch, H. 1997. Satellite tracking of seabirds: interpretation of activity patterns from the frequency of satellite locations. Ibis, 139: 403–405.

Haelters, J., Vigin, L., Stienen, E., Scory, S., Kuijken, E., and Jacques, T. 2004. Importance Ornithologique des espaces marins de la Belgique. Bulletin de L'Institut Royal des Sciences Naturelles de Belgique. Vol 74 Suppl., 90 pp.

Hamer, K., Phillips, R., Wanless, S., Harris, M., and Wood, A. 2000. Foraging ranges, diets and feeding locations of gannets *Morus bassanus* in the North Sea: evidence from satellite telemetry. Marine Ecology Progress Series, 200: 257–264.

Heath, M., and Evans, M. I. 2000. Important Bird Areas in Europe. Priority Sites for Conservation. Vol. I–II. (BirdLife Conservation Series No. 8) BirdLife International, Cambridge.

Hemson, G., Johnson, P., South, A., Kenward, R., Ripley, R., and MacDonald, D. 2005. Are kernels the mustard? Data from global positioning (GPS) collars suggests problems for kernel home-range analyses with least-squares cross-validation. Journal of Animal Ecology, 74: 455–463.

Hunt, G. L.Jr., and Schneider, D. C. 1987. Scale-dependent processes in the physical and biological environment of marine birds. *In* Seabirds: feeding ecology and role in marine ecosystems. Pages 7–41. Ed. By J. P. Croxall. Cambridge University Press, Cambridge.

Hyrenbach, K., Forney, K., and Dayton, P. 2000. Marine protected areas and ocean basin management. Aquatic Conservation: Marine and Freshwater Ecosystems, 10: 437–458.

Jouventin, P., and Weimerskirch, H. 1990. Satellite tracking of Wandering Albatrosses. Nature, 343: 746–748.

Kahlert, J., Desholm, M., Clausager, I., and Petersen, I. K. 2000. Environmental impact assessment of an offshore wind park at Rødsand. Natural Environment Research Institute, Rønde.

Leopold, M. F., Dijkman, E. M., Gonzalez-Mirelis, G., and Berrevoets, C. In press. Marine protected areas in the Dutch sector of the North Sea: a bird's eye view. IMARES/RIKZ Report.

Lindeboom, H., Geurts van Kessel, J., and Berkenbosch, L. 2005. Areas with special ecological values on the Dutch Continental Shelf. Rapport RIKZ/2005.008, Alterra Rapport nr. 1203, 103p.

Louzao, M., Hyrenbach, K. D., Arcos, J. M., Abelló, P., Gil de Sola, L., and Oro, D. 2006. Oceanographic habitat of a critically endangered Mediterranean Procellariiform: implications for the design of Marine Protected Areas. Ecological Applications 16: 1683–1695.

Lindeboom, H., Kessel, J., and Berkenbosch, L. 2005. Areas with special ecological values on the Dutch Continental Shelf. Report RIKZ/2005.008 and Alterra Report nr. 1203, 103 pp.

Madroño, A., González, C., and Atienza, J. C. eds. 2004. Libro Rojo de las Aves de España Dirección General para la Biodiversidad-SEO/BirdLife.

Mañosa, S., Oro, D., and Ruiz, X. 2004. Activity patterns and foraging behaviour of Audouin's gulls in the Ebro Delta, NW Mediterranean. Sci Mar., 68: 605–614.

McSorley, C. A., Dunn, T. E., Gray, C., Reid, J. B., Webb, A., and Dean, B. J. 2005. The spatial extent of Manx shearwater *Puffinus puffinus* evening rafts at Skomer, Rum and Bardsey: Implications for seaward boundary extensions to existing UK qualifying colony Special Protection Areas. JNCC Report, JNCC, Peterborough.

McSorley, C. A., Webb, A., Dean, B. J., and Reid, J. B. 2004. Recommendations for the selection UK inshore Special Protection Areas: a methodological evaluation of site selection and definition of the extent of the interest feature using line transect data. JNCC Report No. 344. JNCC, Peterborough.

Monaghan, P., Walton, P., Wanless, S., Uttley, J. D., and Burns, M. D. 1994. Effects of prey abundance on the foraging behaviour, diving efficiency and time allocation of breeding guillemots *Uria aalge*. Ibis, 136: 214–222.

Ostrand, W. D., Drew, G. S., Suryan, R. M., and McDonald, L. L. 1998. Evaluation of radio-tracking and strip transect methods for determining foraging ranges of Black-legged Kittiwakes. Condor, 100: 709–718.

Pearson, T. H. 1968. The feeding biology of sea-bird species breeding on the Farne Islands, Northumberland. Journal of Animal Ecology, 37: 521–552.

Petersen, I. K., Pihl, S., Hounisen, J. P., Holm, T. E., Clausen, P., Therkildsen, O., and Christensen, T. K. 2006: Landsdækkende optællinger af vandfugle, januar og februar 2004. Danmarks Miljøundersøgelser. 76 s. – Faglig rapport fra DMU nr. 606.

Powell, R. A. 2000. Animal home ranges and territories and home range estimators. *In* Research Techniques in Animal Ecology: Controversies and Consequences, pp.65–110. Ed. by L. Boitani and Fuller, T. K., Columbia University, New York.

RSPB 2004. Report on Birds Directive European Court of Justice cases with case summaries. Ed. by E. Robb, R. Sutherland and Z. Waliczky, RSPB.

RSPB 2005. RSPB SPA Project 2005. [Comparison of IBAs and SPAs in the UK] RSPB CD.

Shealer, D. A. 2002. Foraging behavior and food of seabirds. *In* Biology of marine birds. Pages 137–177, Ed. by E. A. Schreiber and J. Burger, CRC Press, Boca Raton.

Skov, H., Durinck, J., Leopold, M. F., Tasker, M. L. 2007. A quantitative method for evaluating the importance of marine areas for conservation of birds. Biological Conservation, 136: 362–371.

Skov, H., Vaitkus, G., Flensted, K. N., Grishanov, G., Kalamees, A., Kondratyev, A., Leivo, M., Luigujõe, L., Mayr, C., Rasmussen, J. F., Raudonikis, L., Scheller, W., Sidlo, P. O., Stipniece, A., Struwe-Juhl, B., and Welander, B. 2000. Inventory of coastal and marine Important Bird Areas in the Baltic Sea. BirdLife International, Cambridge.

Skov, H., Durinck, J., Leopold, M. F., and Tasker, M. L. 1995. Important bird areas for seabirds in the North Sea including the Channel and the Kattegat. BirdLife International, Cambridge.

SPEA–SEO/BirdLife. 2005. Implementing N2000 in the marine environment. Marine IBAs Lisbon-Vilanova, conclusions.

SPEA-SEO/BirdLife. 2006. Implementing N2000 in the marine environment Marine IBAs: Lisbon-Vilanova, 2006 update. Unpublished document.

Stroud, D. A., Chambers, D., Cook, S., Buxton, N., Fraser, B., Clement, P., Lewis, I., McLean, I., Baker, H., and Whitehead, S. 2001. The UK SPA network: its scope and content. Volumes 1–3. JNCC, Peterborough.

Tucker, G. M., and Evans, M. I. 1997. Habitats for Birds in Europe. A Conservation Strategy for the Wider Environment. (BirdLife Conservation Series No. 6) BirdLife International, Cambridge.

Wanless, S., Bacon, P. J., Harris, M. P., Webb, A. D., Greenstreet, S. P.R., and Webb, A. 1997. Modelling environmental and energetic effects of feeding performance and distribution of shags (*Phalacrocorax aristotelis*): integrating telemetry geographical information systems and modelling techniques. ICES Journal of Marine Science, 54: 524–544.

Wanless, S., Bacon, P. J., Harris, M. P., and Webb, A. D. 2003. Evaluating the coastal environment for marine birds. Eds. Green, D. R., and King, S. D. Coastal and Marine Geo-information Systems, pp. 221–232. Kluwer Academic Publishers, Netherlands.

Webb, A., and Durinck, J. 1992. Counting birds from ship. *In* Manual for aeroplane and ship surveys of waterfowl and seabirds. pp. 24–37. Eds. J. Komdeur, J. Bertelsen, and G. Cracknell), IWRB Special Publication No. 19, Slimbridge.

Webb, A., McSorley, C. A., Dean, B. J., and Reid, J. B. 2004a. Modelling the distribution and abundance of black scoter *Melanitta nigra* in Carmarthen Bay in winter 2001/02: a method for identifying potential boundaries for a marine Special Protection Area. JNCC Report No. 330. JNCC, Peterborough.

Webb, A., McSorley, C. A., Dean, B. J., Reid, J. B., Cranswick, P. A., Smith, L., and Hall, C. 2004b. An assessment of the numbers and distributions of inshore aggregations of waterbirds using Liverpool Bay during the non-breeding season in support of possible SPA identification. Unpublished JNCC Report, JNCC, Peterborough.

Webb, A., McSorley, C. A., Dean, B. J., and Reid, J. B. 2004c. Recommendations for the selection of, and boundary options for, an SPA in Liverpool Bay. Unpublished JNCC Report, JNCC, Peterborough.

Weimerskirch, H., Bonadonna, F., Bailleul, F., Mabille, G., Dell'Omo, G., and Lipp, H.-P. 2002. GPS tracking of foraging Albatrosses. Science, 295: 1259.

Wilson, R. P., Gremillet, D., Syder, J., Kierspel, M. A.M., Garthe, S., Weimerskirch, H., Schäfer-Neth, C., Scolaro, J. A., Bost, C. A., Plötz, J., and Nel, D. 2002. Remote-sensing systems and seabirds: their use, abuse and potential for measuring marine environmental variables. Marine Ecology Progress Series, 228: 241–261.

Wood, A. G., Naef-Daenzer, B., Prince, P. A., and Croxall, J. P. 2000. Quantifying habitat use in satellite-tracked pelagic seabirds: application of kernel estimation to albatross locations. Journal of Avian Biology, 31: 278–286.

## 4       Recommendations for a comprehensive monitoring programme for seabirds

At its 2005 meeting, ICES (2005) recommended the group should perform a review of the variety of methods applied across the North Atlantic region to monitor the performance of seabirds and to assemble a set of standardised and cost-efficient guidelines that could make monitoring more amenable to broad-scale analysis across regions and national borders. Developing recommendations for a comprehensive monitoring programme for seabirds was therefore put up as a term of reference for the 2006 meeting. This chapter was updated at the March 2007 meeting, and includes a summary of the findings of a workshop on seabird population monitoring in Europe, updates the text box on Scottish Seabird monitoring with findings of a recent analysis of productivity data and includes a new section on the use of seabirds as bio-monitors of change in the marine environment.

### 4.1     Introduction

Seabirds are identified as very valuable components of marine ecosystems, not only for their attractiveness and recreational value, but also because many of them have proven to be excellent indicators of important changes in the marine environment (Furness and Monaghan, 1987; Furness and Camphuysen, 1997). Thus, often, seabird data give early indications of fluctuations in fish stocks and oceanographic conditions (Montevecchi, 1993). Monitoring in its broad sense can be defined as the process of gathering information about some system state variables at different points in time for the purpose of assessing system state and drawing inferences about change in state over time (Yoccoz *et al.,* 2001). In the case of monitoring programmes for seabirds, the systems of interest are typically seabird populations, but for practical and logistic reasons we are often limited to monitor the breeding part of the population. The state variables of interest include quantities like breeding population size, reproductive success, diet, pollutant concentration or adult survival.

Monitoring programmes for seabirds have been implemented in European and North American countries for many years. The scale, design and intensity of these programmes vary greatly. Some have been extensive, like for instance the monitoring of seabird breeding numbers and breeding success in Britain and Ireland, which involves yearly surveys of hundreds of sites for several species (Mavor *et al.,* 2005), and thus enables the coverage of large geographic areas. Additionally, some seabird populations have been the subject of more intensive and long-term programmes conducted on a few locations (Wooller *et al.,* 1992). Such monitoring has allowed the detection of effects of various environmental changes on seabird populations at different scales, such as the effect of climate, fisheries and oil spill pollution (e.g. Durant *et al.,* 2003; Frederiksen *et al.,* 2004; Votier *et al.,* 2005). Obviously, the extent and strength of the inferences drawn from these monitoring programmes varies with the design used.

In this context, recommendations for the development of comprehensive monitoring programmes for seabirds can build on the experience gained from these programmes, and can be done by considering three key questions that need to be addressed for any monitoring programme: why monitoring, what to monitor (which species and parameters), and how to monitor (including sampling frequency and how to estimate parameters while accounting for possible sources of error).

### 4.2     Why monitoring?

A critical step in the design of any monitoring programme is to identify the reason why the monitoring programme is to be implemented, as this will determine what parameters should be monitored and how. This is especially important as monitoring activities have often been

criticized for their lack of justification. Sound monitoring programmes can be implemented either to detect effects of environmental impacts or changes, for management purposes (e.g. to detect the need for potential management measures and/or to detect the effect of management measures to keep the system in a given state), or for pure scientific reasons (often it then relates to attempting to understand processes underlying changes in system state). As the understanding of underlying processes is often of direct or indirect relevance for knowing what management measures should be taken, comprehensive monitoring programs could ideally combine scientific and management objectives.

Seabird monitoring provides additional value as the performance of many seabirds indicates important changes in the marine environment. The main focus of the monitoring we consider here is to assess the health of seabird populations as suggested by the EcoQO proposed by ICES (2005). The overall purpose is to detect undesired trends in time to uncover the most important reasons, such that any mitigating actions can be identified and implemented in time to be effective.

In this context, it should be noted that an increased interest has become apparent in spatial issues linked with the response of seabird populations to environmental changes, which has direct implications for the development of monitoring programmes (Box 4−1). This is because such monitoring programmes should enable evaluation of the effect of large scale changes of the environment on seabird populations (e.g. see Chapter 3), but also because the meta-population structure of seabird populations, and their exploitation of large areas of the seas at different time of the year, make them exposed to various factors at different scales.

**Box 4−1. Spatial scale in seabird monitoring.**

One of the issues in the design of monitoring programmes is the selection of representative colonies. This requires some assumptions or preferably knowledge about the spatial scale of any geographical patterns in monitored parameters. At the extremes, if all colonies behave differently, it is not possible to select representative colonies and, if there is no or little spatial variation, purely practical considerations will determine which colonies to monitor. More realistically, there will be some spatial structuring in most or all monitored parameters, but little information about the most relevant scale to monitor will usually be available when a programme is being designed. However, existing knowledge about relevant aspects of the physical or biological environment can often provide useful pointers for the regional structure of the programme, together with current knowledge of the population biology of the species considered.

Spatial variation in demographic parameters is not well known for most seabirds, but some interesting studies have been carried out on black-legged kittiwakes and Atlantic puffins (e.g., Frederiksen *et al.*, 2005a, 2005b; Harris *et al.*, 2005). On the wide-range scale, the black-legged kittiwake has extremely variable survival and reproductive success, with colonies in the N Pacific having higher survival and lower reproductive success than most Atlantic colonies (Frederiksen *et al.,* 2005a). At the smallest scale, survival and reproductive success can vary between nearby colonies or parts of the same colony (Danchin and Monnat 1992), and change in local numbers of breeders can be largely explained by differential dispersal and recruitment of individuals among colonies or sub-colonies (Danchin *et al.,* 1998). On intermediate scales, possibly more relevant for the design of monitoring programmes, analyses of data from the UK Seabird Monitoring Programme have shown clear evidence of spatial structuring, with reproductive success being highly correlated between colonies within regions, but mostly uncorrelated between regions (Frederiksen *et al.* 2005b). Further analyses show that reproductive success at seven colonies within one such region, SE Scotland, all show similar temporal patterns and relationships with environmental parameters, and that one colony is particularly representative of the region as a whole, with a correlation coefficient of 0.96 between this colony and the regional mean (Frederiksen *et al.,* in prep.). Further work is needed to confirm whether these findings can be generalised to other regions and species, and thus whether it is generally possible to designate biologically meaningful regions for monitoring as well as representative colonies in each region. Further work on factors affecting dispersal and recruitment at various hierarchical scales could be especially interesting in these respects, as it could help quantifying how independent nearby colonies are in terms of breeding numbers.

## 4.3   Ongoing seabird monitoring in the ICES areas

Existing monitoring programmes vary in many ways and in order to fully review the current monitoring of seabirds in the ICES region, WGSE decided to distribute a questionnaire (Annex 5 in the 2006 report) to all countries to map what their present monitoring schemes cover in terms of species, sites, parameters and monitoring frequencies. The aim was to present a summary and more detailed review of this material in the present report to provide important basic elements for evaluating the possibility of actually developing large scale comprehensive programmes.

However, unknown to the WGSE at that time, a workshop on European seabird monitoring to be held in Aberdeen in September 2006 was being planned, prior to which a similar questionnaire was to be circulated to representatives of 34 countries to determine what

breeding seabird databases existed, the temporal and spatial extent of seabird monitoring in Europe and to perform a 'stock-take' of what information exists and how representative it is. A summary of this workshop is given in Box 4–2. It was decided that this questionnaire would replace that planned by the WGSE, and, at the September workshop, the results were summarized as follows: *18 countries returned completed questionnaires. 15 out of 18 had count data in a time series back to at least 1970s and 1980s (mostly). Of these, 80% monitored some species annually and 69% now have standardised methods. Countries with a small number of species (and colonies) tended to regularly monitor a much larger proportion of colonies and therefore potentially produce more accurate short-term trends than countries with more seabirds, where much smaller proportions of colonies could be monitored annually. Some countries with large numbers of seabirds had, however, conducted more than one census in the past and so could accurately assess long-term trends. Ten countries also collected productivity data but fewer countries collected other data such as diet or adult survival. A need was expressed to obtain information from those countries that have not yet responded to the questionnaire before carrying out a more in depth investigation of precisely how much of the count data in each country will stand up to time-series analysis. A further workshop is planned at the European Bird Census Council (EBCC) conference in spring 2007 to provide such information.* A brief summary of the main points of the questionnaire is given in Tables 4.1 and 4.2.

The original scope of the WGSE questionnaire included the entire North Atlantic region, including the New World (USA, Canada, Greenland), and in addition to the European countries which did not respond, there is also need to obtain similar information from USA and Canada. Another important caveat which needs to be filled is details of winter population assessments which were not considered in the questionnaire sent out. Further contact with coordinators of existing schemes is also needed to provide an overview of the current status, population trends and demographics of seabirds within the entire North Atlantic region in order to be able formulate concrete (methodological) recommendations, and to stimulate national or regional authorities to (re-)consider monitoring schemes to be established to fill in major gaps.

**Table 4.1. List of seabird population databases in Europe. Cells left blank are where availability of data is unknown.**

| COUNTRY | IS THERE A BREEDING SEABIRD DATABASE? | APPROX. NUMBER OF COLONIES | FORMAT | DATA AVAILABLE? | | | | | OTHER |
|---|---|---|---|---|---|---|---|---|---|
| | | | | BREEDING SEABIRD COUNTS | PRODUCTIVITY | DIET | FEEDING RATES | PHENOLOGY | |
| Greenland | Yes | >1000 | database | YES | | | | | |
| Norway | Yes | 101–1000 | database | YES | YES | YES | YES | YES | Adult survival |
| Faeroes | Yes | >1000 | database | YES | YES | | | | |
| Iceland | Yes | >1000 | database | YES | YES | | | | |
| Spain | Yes | 11–100 | | YES | YES | YES | | YES | Population dynamics |
| France | Yes | >1000 | database | YES | | | | | |
| Netherlands | Yes | >1000 | | YES | | | | | |
| UK | Yes | >1000 | database | YES | YES | YES | YES | YES | Adult survival |
| Estonia | No | | | | | | | | |
| Finland | Yes | >1000 | spreadsheet | YES | | | | | |
| Latvia | Yes | 101–1000 | spreadsheet | YES | YES | | | YES | |
| Gibraltar | No | 1 to 10 | | YES | | | | | Counts of seabirds on passage |
| Greece | Yes | 11–100 | database | YES | YES | YES | YES | YES | Pollutants |
| Italy | Yes | 11–100 | database | YES | | | | | |
| Malta | Yes | 11–100 | database | YES | YES | YES | | YES | Human impacts |
| Montenegro | No | | | | | | | | |
| Belgium | Yes | 1 to 10 | spreadsheet | YES | YES | YES | YES | NO | |
| Germany | Yes | 101–1000 | spreadsheet | YES | | | | | |
| Republic of Ireland | Yes | >1000 | database | YES | YES | NO | NO | NO | |

**Table 4.2. Methodology and geographic coverage of current monitoring programmes in Europe as of September 2006.**

| COUNTRY | NUMBER OF SPECIES | HOW ARE COLONIES COUNTED?[2] | HAVE ALL COLONIES BEEN SURVEYED AT LEAST ONCE AS PART OF NATIONAL CENSUS? | PROPORTION OF SEABIRD COLONIES SURVEYED IN A GIVEN YEAR?[3] | YEAR OF EARLIEST RECORD | YEAR OF MOST RECENT RECORD | NO. OF SPECIES | GEOGR. COVERAG4[3] | APPROX. NUMBER OF COLONIES | APPROX. PROPORTION OF SEABIRD COLONIES MONITORED | YEAR OF EARLIEST RECORD | YEAR OF MOST RECENT RECORD |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | POPULATION MONITORING | | | | | | PRODUCTIVITY MONITORING[1] | | | |
| Greenland | 22 | CP | No | 1–25% | 1920 | 2006 | | | | | | |
| Norway | 17 | CP | No | 1–25% | 1946 | 2006 | 13 | S | 1 to 10 | | 1942 | 1974 |
| Faeroes | 4 | CP | No | 1–25% | 1972 | 2006 | 1 | S | 1 to 10 | | 1958 | 1982 |
| Iceland | 23 | CP | No | 1–25% | 1840 | 2006 | 3 | S | 11–100 | 1–25% | 1942 | 1974 |
| Spain | 6 | CP | Yes | 26–50% | 1981 | 2006 | 3 | C | 11–100 | 1–25% | 1988 | 2006 |
| France | 26 | C | Yes | 1–25% | 1968 | 2001 | | S | | | | |
| Netherlands | 12 | CP | Yes | 76–100% | 1900 | 2006 | 8 | S | 101–1000 | | 1914 | 1960 |
| UK | 26 | CP | Yes | 1–25% | 1969 | 2006 | 25 | C | 101–1000 | | 1964 | 1985 |
| Finland | 28 | CP | No | 51–75% | 1984 | 2006 | 1 | S | 1 to 10 | | 1952 | 1974 |
| Latvia | 6 | CP | Yes | 1–25% | 1986 | 2002 | | | | | | |
| Gibraltar | 2 | P | Yes | 76–100% | 2002 | 2002 | 1 | S | 11–100 | 1–25% | 1991 | 1997 |
| Greece | 2 | CP | Yes | 26–50% | 1995 | 2006 | | | | | | |
| Italy | 5 | C | Yes | 76–100% | 1980 | 2006 | 4 | C | 11–100 | 76–100% | 1960 | 1983 |
| Malta | 4 | CP | Yes | 76–100% | 1968 | 2006 | 3 | S | 1 to 10 | 76–100% | 1988 | 1997 |
| Belgium | 5 | C | Yes | | 1959 | 2006 | | S | 1 to 10 | 1–25% | | |
| Germany | 20 | CP | Yes | 76–100% | 1991 | 2003 | 8 | S | 11–100 | | 1966 | 1986 |
| Republic of Ireland | 23 | C | Yes | 1–25% | 1969 | 2006 | | | | | | |

[1] All sites monitored are chosen non-randomly, except in Malta where production is monitored in all colonies

[2] P=plots, C=whole colony, CP=both

[3] Apart from during national censuses

[4] S=confined to specific regions, C=representative of entire country

**Box 4−2. A summary of the JNCC-Birdlife European Seabird Indicator workshop.**

Coincident with the 9th International Conference of the Seabird Group in early September 2006, a half-day workshop was arranged in Aberdeen by JNCC, RSPB and BirdLife International. The workshop, which attracted twenty people from ten different European countries, aimed at exploring the possibilities and interest of producing a pan-European Seabird Indicator. This option was discussed at the end of the workshop on the background of nine opening talks covering the following relevant topics: the usefulness of indicators (Mark Eaton, RSPB), the rationales for a pan-European seabird indicator (Ian Burfield, BirdLife International), the work on a Scottish indicator for breeding seabirds (Matt Parsons, JNCC), the selection of suitable monitoring parameters (Norman Ratcliffe, RSPB), the availability of relevant data (Ian Mitchell, JNCC), the EcoQO work by WGSE on how to develop indices of seabird community health (Stefan Garthe, Univ. Kiel), and the international initiatives to collate seabird data such as the Nordic seabird colony database (Ian Mitchell *et al.*), the work made by the MEDMARAVIS (Mediterranean Marine Avifauna) association (John Borg, BirdLife Malta) and the ESAS (European Seabirds at Sea) database (Andy Webb, JNCC).

The workshop attendants, with support from a number of other key individuals that had commented on the concept, considered it worthwhile to pursuing the idea of developing a European Seabird Indicator. As a first step, it was recommended to seek funding to arrange a more intensive, follow-up workshop engaging the co-ordinators of the most comprehensive monitoring programmes and databases that aims to:

1 ) bring various national seabird datasets together in order to explore what species and geographical area could be represented by an indicator (or a set of indicators),

2 ) give recommendations on how the data might be best organised and analysed (e.g. to produce state and/or pressure indicators),

3 ) what targets could be set,

4 ) identify short- and long-term resource requirements and possible sources of funding, and

5 ) propose a long-term strategy and work plan to complete the task.

Such an indicator could form part of the current initiative on Streamlining European 2010 Biodiversity Indicators (SEBI 2010) and serve as a valuable tool for communicating and managing the state of seabird populations across Europe. It would probably also make a valuable contribution to the development of an OSPAR EcoQO for seabirds.

An example of a national seabird monitoring programme is given in Box 4–3 for Scotland. This explains how *ad hoc* sampling of colonies by a network of volunteers can yield robust estimates of regional and time-dependent variation in numbers and productivity provided appropriate statistical methods are employed. However, improved accuracy and precision for a given level of effort will result if surveys are designed appropriately, though the financial costs of surveying randomly selected colonies where no volunteer assistance is available may offset man-hour savings.

**Box 4−3. Seabird monitoring in Scotland.**

The population status and trends of seabirds in Scotland have been monitored using two complementary programmes since 1985. There have been two broadly comprehensive censuses during 1985-87 (Seabird Colony Register, SCR) and 1998-2002 (Seabird 2000). These give snapshot estimates of status and long-term change without colony sampling bias. Between these censuses, counts of whole sample colonies or counts of study plots within colonies have been collated by the Seabird Monitoring Programme (SMP) with the aim of describing annual patterns of change. Productivity data are also collected at a sample of sites as part of the SMP. The complete censuses were carefully designed and co-ordinated, but counts and productivity estimates contributed to the SMP were largely collected on an *ad hoc* basis by professional biologists, reserve wardens and volunteers.

Since data collection for the SMP is not based on a stratified random design, they have many biases that need to be overcome by design of the statistical analysis. Not all sites in all years are sampled, and so missing values have to be imputed to obtain likely trends. Some regions are undersampled relative to the proportion of the national population they host, and so values have to be weighted accordingly in order to avoid bias. Large colonies are less likely to be counted in their entirety than small ones owing to logistics, and this can result in overestimation of population growth rates due to density dependence. Plot counts are often available between complete counts for some species at large colonies, and in order to include these data models have to have a hierarchical design.

Analysis of the count data for selected species in Scotland between 1986 and 2004 was approached using a Bayesian inference model (JNCC 2006) and those for productivity using a generalised linear mixed model (JNCC, in prep.). The models impute missing counts based on trends within and across colonies, have a hierarchical design which allows inclusion of both whole colony and plot counts and weight values by colony/sample size and regional importance to overcome the previously mentioned sampling biases. These approaches are more flexible and there is greater control over the assumptions that in previously employed analytical methods. The hierarchical component of the Bayesian model assumes that plot counts are representative of those across the whole colony, though in some cases this was clearly untrue (e.g. great skuas on Hoy, Orkney).

Despite this problem, the Bayesian model produced accurate trends for most seabird species studied, with extrapolations of annual change from the SCR census predicting status during the Seabird 2000 census reasonably well. The predicted trends were more accurate than those produced by chain indices that suffered from biased sampling with respect to colony size and density-dependent growth. However, in the case of tern species and great cormorant, trends were inaccurate owing to site colonisation and extinction events that the model predictions were unable to track because of assumptions that were made. Although chain indices produced more accurate trends in these species, these remain unsatisfactory, and require further refinement to yield properly weighted estimates of trends with confidence limits. The mixed models produced robust time- and region-dependent estimates of productivity.

The project has demonstrated that data collected on an *ad hoc* basis can yield reliable annual population trends, but that sophisticated analyses are required to overcome the inherent biases in such data. However, in some instances, biases may be extreme and unquantifiable, and so careful design of sampling is advisable to ensure that estimated trends are accurate.

## 4.4 Monitoring priorities

### 4.4.1 Selection of species

A number of specific considerations should be made when selecting the target species for a reasonable effort of seabird monitoring. WGSE recommends that priority is given to those species that are:

- valuable indicators (in terms of their sensitivity, cost-efficiency and early-warning capacity) of ecosystem changes that are important for the well-being and management of other marine resources.

- monitored throughout most of their breeding range within the ICES region, in order to facilitate wide-scale analyses of population dynamics.

- of special conservation concern, either because they are listed on the national and/or European red lists, or they are considered as problem or key species. However, recognition of conservation concern is dependent on monitoring of all species, and so effort should not be directed exclusively to species of current conservation concern otherwise future declines of other species may go undetected. Rather, this criterion should be used to allocate relative monitoring effort among species in an adaptive manner.

- particularly vulnerable to impact factors that are expected to be of extra importance.

- of special international importance, i.e. the national population constitutes a large part (≥25%) of the biogeographical population (normally the European or NE Atlantic population) it belongs to.

Additionally, considerations should also be made with respect to:

- including representatives from each of the main ecological groups of seabirds present as defined by their main feeding areas and feeding ecology (Table 4.3).

- selecting species that have proven to be (or are expected to be) representative for several other seabird species that are more difficult to monitor.

- avoiding species that are highly impractical or unfeasible to monitor properly (e.g. nocturnal, burrow nesting petrels). However, where these score especially highly on the previous list of criteria (e.g. of high conservation concern or national importance), efforts should be made to include such species in monitoring programmes.

**Table 4.3. The seabird species or seabird taxa that occur in significant numbers in European waters, grouped according to their main feeding areas (pelagic versus near-shore or deep versus shallow waters) and feeding behaviour (diving, plunge-diving or surface-feeding) in the breeding season.**

|  | PELAGIC/DEEP WATER | NEAR-SHORE/SHALLOW WATER |
|---|---|---|
| Diving | *Uria* guillemots, Razorbill, Atlantic puffin, Little auk | Divers, grebes, cormorants, seaducks, Black guillemot |
| Plunge-diving | Northern gannet | Terns |
| Surface-feeding | Northern fulmar, petrels, shearwaters, storm-petrels, Black-legged kittiwake | Pelicans, phalaropes, skuas, *Larus* gulls, Ivory gull |

### 4.4.2 Selection of monitoring parameters

ICES (2005) concluded that the huge variety of possible mechanisms underlying changes in seabird breeding numbers makes it necessary to also monitor different parameters of population dynamics for (at least some of) the key species, although these parameters are usually more labour intensive to monitor adequately. The great advantage of this approach is that it enables an immediate exploration of possible reasons for any population trends of

special concern, without having to spend additional years to collect less adequate *a posteriori* information on the same parameters. As most seabirds are migratory outside the breeding season, trends in population numbers might also be affected by environmental conditions far away from the breeding areas, in many cases outside the ICES areas in question. These factors are probably best reflected by changes in adult survival rates, whereas reproductive rates and chick diets (as well as other aspects of adults' feeding ecology or their physical condition) are likely to be better indicators of local conditions within the breeding seasons.

Based on this, WGSE recommends that for all breeding species identified as important targets for monitoring, the monitoring should preferably produce series of annual data for the following key parameters:

- Population size
- Adult survival rate
- Reproductive success (specific parameters varying between species)
- Diet of breeding adults and/or chicks

Whereas the first three are needed to explore the essential dynamics of the target populations, data on diet are considered valuable as many environmental factors (man-induced as well as natural variation) affect seabirds indirectly through their food base. Except for the sampling of diets, which is treated in more detail in Chapter 6, references to standardised methods for sampling these parameters are given in Section 4.5.4. Due to the relatively high efforts needed to carry out state-of-the-art monitoring of survival rates (by capture-recapture techniques) and to sample diets throughout a significant part of the breeding period, these parameters are usually best monitored on a limited selection of sites, only.

Members of the WGSE have produced several reviews on the value and suitability of different population parameters for monitoring purposes (e.g. Becker and Chapdelaine, 2003; Furness *et al.,* 2003). For example, parallel to the collection of data on the key parameters listed above, additional and potentially valuable information might be collected with little extra effort, including simple indices of foraging performance (e.g. feeding rates and breeding site attendance), breeding phenology and likelihood of breeding, as well as sampling tissues for various purposes (e.g. contaminant levels, parasites and pathogens, genetic and stable isotope analysis, sexing).

## 4.5 Monitoring methods

### 4.5.1 Hierarchical approach combining utility and practicality

Depending on the objectives of a given monitoring programme and the parameters to be monitored, some hierarchical approach combining an extensive survey of numbers of breeders, productivity and chick diet on samples of study plots at different locations and more intensive monitoring of annual survival in a limited number of locations could be suggested.

In addition to practical aspects of monitoring methods, sound monitoring has to consider potential sources of errors when estimating parameters for seabird populations. Of these, detection errors and spatial variability, are discussed in the next Section (4.5.2), and a more specific discussion of sampling and analytical design for population trend estimation is provided thereafter (Section 4.5.3).

### 4.5.2 Accounting for sources of error: detectability issues and spatial variability.

Two important sources of variability of parameter estimates of seabird populations are detection error and survey error in relation to spatial variability.

For breeding numbers and demographic parameters such as survival, the first source of error occurs because few survey methods permit the detection of all individual animals or breeding events in surveyed areas (Williams *et al.,* 2001). For estimating parameters like annual survival rate, capture-mark-recapture methodology is now commonly used for seabird populations (Cam *et al.,* 1998; Frederiksen *et al.,* 2004; Oro *et al.,* 2004; Harris *et al.,* 2005a, Sandvik *et al.,* 2005) and is highly recommended. Classically, this approach involves the marking of individuals in the field, their recapture at later occasions and the use of probabilistic modelling to account for the fact that some individuals remain undetected in the field even when they are still alive (Lebreton *et al.,* 1992). One important limit to this sort of approach is that individuals are considered dead if they leave the study area, which thus confounds permanent emigration with mortality (this is why the term "apparent survival rates" is used). The assumptions made in terms of homogeneity in detection probabilities among individuals are also important to consider and various approaches have been proposed to account for these issues, usually combining complementary sources of information.

The second source of error of population parameter estimation involves the inability to survey large areas entirely, and the resulting need to draw inference about large areas based on (usually non-random) samples of locations within those areas. What should be avoided is the focus on a few very subjectively chosen monitoring plots and the sampling design should provide a survey of a representative set of monitoring plots. As illustrated in Box 4−1, these issues can be especially important at some small spatial scales. The next section explains how the sampling and analytical design of seabird population trends can allow dealing with some of these issues.

### 4.5.3   Sampling and analytical design for population trend estimation

Ideally, data on numbers and productivity would be available annually for all colonies of a species of interest. This is achieved for some relatively rare or restricted range species (e.g. Roseate Tern) but for most species, the number and size of colonies are too great for this to be practicable. In these instances, sampling through time and space is needed to estimate variation in numbers and productivity, and surveys and/or data analyses have to be carefully designed in order to avoid bias.

Complete censuses overcome spatial sampling bias and provide information and absolute status, distribution and relative importance of regions or colonies (Mitchell *et al.,* 2004). Complete censuses are complementary to annual sample surveys as they facilitate assessment of potential sampling biases when designing data collection or analyses, and independent *post-hoc* assessment of trend accuracy (JNCC, 2006). Repeat censuses also provide information on population change but, owing to the long time intervals between these, distinguishing longer-term population trends from shorter-term fluctuations can be difficult or take several decades to become evident.

Annual population change and productivity can be estimated by sampling counts at the whole colony or study plot scale on a year-to-year basis (e.g. Mavor *et al.*, 2005). Ideally, the colonies and plots would be selected according to a random stratified design to avoid bias in trends, and each site would be counted every year. In most countries, however, annual sampling is on an *ad hoc* basis by a mixture of professional biologists, wardens and volunteers. These data have the potential to produce accurate trends for minimal cost, provided that inherent biases are overcome analytically.

In most *ad hoc* surveys, not all colonies are counted in every year, such that comparisons of summed annual counts reflects both changes in status and the pattern of missing counts (Ter Braak *et al*., 1994). Spatial bias is also common in *ad hoc* surveys, with those colonies or plots that are easy to access or observe being over-sampled. Bias can also occur relative to density. Density-dependence often results in large colonies growing more slowly than small ones, and

in core areas of colonies changing more slowly than peripheral ones (e.g. Moss *et al.*, 2002). Whole colony counts are often biased towards small colonies as they are easier to count annually, with the result that colony growth rates are overestimated. Sometimes plots are initially located in areas of the colony where birds are breeding and thus do not capture expansion into areas that were unused when the plots were set up, further underestimating rates of increase. Plot counts are often biased towards core areas where there are large numbers of birds available to count, but since these have often reached their maximum capacity population growth for the colony as a whole may be underestimated. Furthermore, plot counts are often discontinued when the number of birds becomes "too few to be worth counting", which results in rates of decline being underestimated also. Productivity values are ratios, and so need to be weighted according to sample size to prevent events at small colonies having an unduly large influence on overall patterns of change.

Biases in *ad hoc* annual surveys can be overcome by modelling in some cases. Problems with missing values can be remedied using chain indices, in which only colonies counted in consecutive years are included in analysis, but this approach is wasteful of data and may exacerbate other biases (Ter Braak *et al.*, 1994). GLMs such as TRIM (Pannekoek and van Strien, 2001) or Bayesian inference models (JNCC, 2006) are preferable to impute missing counts, and these also allow statistical significance of trends or year-to-year changes to be tested. Of these, the Bayesian method is the most flexible as it allows both whole colony and plot counts to be modelled in a hierarchical manner and makes weaker assumptions regarding synchronicity of trends across sites compared to GLMs (which reduces density-related biases). Productivity ratios can be weighted according to sample size, or counts of chicks fledged can be modelled with sample sizes included as a denominator or offset to ensure appropriate weighting of values according to sample size. Spatial bias in values can be overcome by weighting trends according to regional status as determined from complete census data where this is available (JNCC 2006, in prep). An example from modelling seabird trends in Scotland is given in Box 4−3. Sophisticated analyses cannot be relied on to overcome shortcomings in all data however, and surveys should be designed to minimise the biases discussed above in order to attain the maximum accuracy.

The precision of trends as well as their accuracy is a consideration when designing surveys. Sampling has to be at an effort that allows trends of interest to be detected within an acceptable time-period. The statistical power to detect trends increases with sample size but declines with variance in trends among sampled units (Steidl and Thomas, 2001). Power analyses can be used to determine the minimum sample size needed to detect the desired rate of change for an observed level of variance within an acceptable time-period and these can be useful to produce a parsimonious monitoring programme (Anker-Nilssen *et al.*, 1996; Sims *et al.*, in press).

When designing monitoring programmes the dynamic nature of seabird populations needs to be considered. Shifts in range or distribution may result in biases changing, and increases in variability of trends among units may result in loss of power. Monitoring programmes therefore need to be reviewed periodically to maintain the desired accuracy and precision of results.

### 4.5.4 References to descriptions of standardised monitoring methods

Standardised methods for monitoring of population size and reproductive performance are described for a variety of seabird species by Walsh *et al.* (1995). In relation to the challenges outlined in the above sections, we aim at reviewing these and supplementary methods in more detail at our next meeting.

Standard methods for designing capture-mark-recapture programme to estimate a key demographic parameter like annual survival rate are described in various outlets (Pollock *et*

*al.,* 1990; Lebreton *et al.,* 1992; Williams *et al.,* 2002), available together with software for data analyses (e.g. White and Burnham, 1999). As seen in Section 4.5.2, the application of such approaches to seabird populations has been developing greatly over the last 10 years, and the capture-mark-recapture of individuals is now a significant component of most colony-based monitoring programmes of seabird populations.

## 4.6 Seabirds as biomonitors

In this chapter we have reviewed some ways in which it is possible to monitor aspects of seabird population numbers and ecology in order to understand how seabird populations are being affected by changes in the environment. The principal aim is to be able to inform management that will conserve seabirds. It is also the case that seabirds can be used as a tool to monitor a number of aspects of the marine environment, such as levels of pollution. Examples of this include the monitoring of levels of oil pollution by beached bird surveys (Camphuysen and Heubeck 2001; Seys *et al.,* 2002a, 2002b; Roletto *et al.,* 2003; Camphuysen *et al.,* 2005), levels of plastic on the ocean surface by analysis of seabird stomach contents (Spear *et al.,* 1995; Blight and Burger 1997; Cadee 2002; Anon, 2003; Guse *et al.,* 2005; van Franeker *et al.,* 2005), levels of mercury in marine food webs by analysis of selected feathers of seabirds (Furness *et al.,* 1995; Monteiro and Furness 1995; Goutner *et al.,* 2000; Bearhop *et al.,* 2000a, 2000b; Arcos *et al.,* 2002; Burgess *et al.,* 2005; Scheifler *et al.,* 2005; Tavares *et al.,* 2005; Champoux *et al.,* 2006), and levels of organochlorines in seabird eggs as a measure of both temporal trends and spatial pattern (Denker *et al.,* 1994; Thyen *et al.,* 2000; Albanis *et al.,* 2003; Bustnes *et al.,* 2003; Munoz Cifuentes *et al.,* 2003; Bustnes *et al.,* 2005; Harris *et al.,* 2005b; Helberg *et al.,* 2005; Lundstedt-Enkel *et al.,* 2005; Verreault *et al.,* 2005). These uses are well established, and some others are still at an early stage of development. However, we have not reviewed the uses of seabirds as monitors of the wider environment in this paper, but have confined our review only to the monitoring of seabird populations and demography as a tool to understand seabird population changes. We are however confident that monitoring the trends, demographics and diets of a reasonable selection of seabird populations identified according to the criteria proposed in Section 4.4.1, will provide useful information for indicating important changes in other components of the marine environment.

## 4.7 References

Albanis, T. A., Goutner, V., Konstantinou, I. K., and Frigis, K. 2003. Organochlorine contaminants in eggs of the yellow-legged gull (*Larus cachinnans michahellis*) in the North Eastern Mediterranean: is this gull a suitable biomonitor for the region? Environmental Pollution, 126: 244–255.

Anker-Nilssen, T., Erikstad, K. E., and Lorentsen, S.-H. 1996. Aims and effort in seabird monitoring: an assessment based on Norwegian data. Wildlife Biology, 2: 17–26.

Anon. 2003. Ecological quality objectives. ICES Cooperative Research Report, 262: 40–116.

Arcos, J. M., Ruiz, X., Bearhop, S., and Furness, R. W. 2002. Mercury levels in seabirds and their fish prey at the Ebro Delta (NW Mediterranean): the role of trawler discards as a source of contamination. Marine Ecology Progress Series, 232: 281–290.

Bearhop, S., Waldron, S., Thompson, D., and Furness, R. W. 2000a. Bioamplification of mercury in great skua *Catharacta skua* chicks: the influence of trophic status as determined by stable isotope signatures of blood and feathers. Marine Pollution Bulletin, 40: 181–185.

Bearhop, S., Phillips, R. A., Thompson, D. R., Waldron, S., and Furness, R. W. 2000b. Variability in mercury concentrations of great skuas *Catharacta skua*: the influence of colony, diet and trophic status inferred from stable isotope signatures. Marine Ecology Progress Series, 195: 261–268.

Becker, P. H., and Chapdelaine, G. 2003. Further development of seabird monitoring. ICES Cooperative Research Report, 258: 52–60.

Blight, L. K., and Burger, A. E. 1997. Occurrence of plastic particles in seabirds from the eastern North Pacific. Marine Pollution Bulletin, 34: 323–325.

Burgess, N. M., Evers, D. C., and Kaplan, J. D. 2005. Mercury and other contaminants in common loons breeding in Atlantic Canada. Ecotoxicology, 14: 241–252.

Bustnes, J. O., Erikstad, K. E., Skaare, J. U., Bakken, V., and Mehlum, F. 2003. Ecological effects of organochlorine pollutants in the arctic: A study of the glaucous gull. Ecological Applications, 13: 504–515.

Bustnes, J. O., Miland, O., Fjeld, M., Erikstad, K. E., and Skaare, J. U. 2005. Relationships between ecological variables and four organochlorine pollutants in an arctic glaucous gull (*Larus hyperboreus*) population. Environmental Pollution, 136: 175–185.

Cadee, G. C. 2002. Seabirds and floating plastic debris. Marine Pollution Bulletin, 44: 1294–1295.

Cam, E., Hines, J. E., Monnat, J.-Y., Nichols, J. D., and Danchin, E. 1998. Are adult non-breeders prudent parents? The kittiwake model. Ecology, 79: 2917–2930.

Camphuysen, C. J., and Heubeck, M. 2001. Marine oil pollution and beached bird surveys: the development of a sensitive monitoring instrument. Environmental Pollution, 112: 443–461.

Camphuysen, C. J., Fleet, D. M., Reineking, B., and Skov, H. 2005. Oil pollution and seabirds. Wadden Sea Ecosystem, 19: 115–122.

Champoux, L., Masse, D. C., Evers, D., Lane, O. P., Plante, M., and Timmermans, S. T.A. 2006. Assessment of mercury exposure and potential effects on common loons (*Gavia immer*) in Quebec. Hydrobiologia, 567: 263–274.

Danchin, E., and Monnat, J.-Y. 1992. Population dynamics modelling of two neighbouring Kittiwake *Rissa tridactyla* colonies. Ardea, 80: 171–180.

Danchin, E., Boulinier, T., and Massot, M. 1998. Conspecific reproductive success and breeding habitat selection: implications for the study of coloniality. Ecology, 79: 2415–2428.

Denker, E., Becker, P. H., Beyerbach, M., Buthe, A., Heidmann, W. A., and Staats de Yanes, G. 1994. Concentrations and metabolism of PCBs in eggs of waterbirds on the German North Sea coast. Bulletin of Environmental Contamination and Toxicology, 52: 220–225.

Durant, J. M., Anker-Nilssen, T., and Stenseth, N. C. 2003. Trophic interactions under climate fluctuations: the Atlantic puffin as an example. Proceedings of the Royal Society of London series B, 270: 1461–1466.

Frederiksen, M., Harris, M. P., and Wanless, S. 2005a. Inter-population variation in demographic parameters: a neglected subject? Oikos, 111: 209–214.

Frederiksen, M., Wanless, S., Harris, M. P., Rothery, P., and Wilson, L. J. 2004. The role of industrial fisheries and oceanographic change in the decline of North Sea black-legged kittiwakes. Journal of Applied Ecology, 41: 1129–1139.

Frederiksen, M., Wright, P. J., Heubeck, M., Harris, M. P., Mavor, R. A., and Wanless, S. 2005b. Regional patterns of kittiwake *Rissa tridactyla* breeding success are related to variability in sandeel recruitment. Marine Ecology Progress Series, 300: 201–211.

Furness, R. W., and Camphuysen, C. J. 1997. Seabirds as monitors of the marine environment. ICES Journal of Marine Science, 54: 726–737.

Furness, R. W., and Monaghan, P. 1987. Seabird ecology. Blackie, Glasgow.

Furness, R. W., Becker, P. H., Hüppop, O., and Davoren, G. 2003. Review of the sensitivity of seabird populations to changes in life history parameters. ICES Cooperative Research Report, 258: 26–36.

Furness, R. W., Thompson, D. R., and Becker, P. H. 1995. Spatial and temporal variation in mercury contamination of seabirds in the North Sea. Helgoländer Meeresuntersuchungen, 49: 605–615.

Goutner, V., Furness, R. W., and Papakonstantinou, K. 2000. Mercury in feathers of Audouin's gull (*Larus audouinii*) chicks from northeastern Mediterranean colonies. Archives of Environmental Contamination and Toxicology, 39: 200–204.

Guse, N., Fleet, D., van Franeker, J., and Garthe, S. 2005. Der Eissturmvogel (*Fulmarus glacialis*)–Mülleimer der Nordsee? Seevögel, 26: 3–12.

Harris, M. P., Anker-Nilssen, T., McCleery, R. H., Erikstad, K. E., Shaw, D. N., and Grosbois, V. 2005a. Effect of wintering area and climate on the survival of adult Atlantic puffins *Fratercula arctica* in the eastern Atlantic. Marine Ecology Progress Series, 297: 283–296.

Harris, M. L., Wilson, L. K., and Elliott, J. E. 2005b. An assessment of PCBs and OC pesticides in eggs of double-crested (*Phalacrocorax auritus*) and pelagic (*P. pelagicus*) cormorants from the west coast of Canada, 1970 to 2002. Ecotoxicology, 14: 607–625.

Helberg, M., Bustnes, J. O., Erikstad, K. E., Kristiansen, K. O., and Skaare, J. U. 2005. Relationships between reproductive performance and organochlorine contaminants in great black-backed gulls (*Larus marinus*). Environmental Pollution, 134: 475–483.

ICES. 2005. Report of the Working Group on Seabird Ecology 2005. ICES CM 2005/G:07. 49 pp.

JNCC. 2006. Natural Heritage Trends: Abundance of breeding seabirds in Scotland. Scottish Natural Heritage Report No F05NB01, Edinburgh, UK.

Lebreton, J.-D., Burnham, K. P., Clobert, J., and Anderson, D. R. 1992. Modeling survival and testing biological hypotheses using marked animals: a unified approach with case studies. Ecological Monographs, 62: 67–118.

Lundstedt-Enkel, K., Tysklind, M., Trygg, J., Schueller, P., Asplund, L. *et al.* 2005. A statistical resampling method to calculate biomagnification factors exemplified with organochlorine data from herring (*Clupea harengus*) muscle and guillemot (*Uria aalge*) egg from the Baltic Sea. Environmental Science & Technology, 39: 8395–8402.

Mavor, R. A., Parsons, M., Heubeck, M., and Schmitt, S. 2005. Seabird numbers and breeding success in Britain and Ireland, 2004. UK Nature Conservation Report No. 29. Joint Nature Conservation Committee, 104 pp.

Monteiro, L. R., and Furness, R. W. 1995. Seabirds as monitors of mercury in the marine environment. Water Air and Soil Pollution, 80: 851–870.

Montevecchi, W. A. 1993. Birds as indicators of change in marine prey stocks. In: Furness, R. W., and Greenwood, J. J.D. eds. Birds as monitors of environmental change. Chapman & Hall, London: 219–266.

Moss, R., Wanless, S., and Harris, M. P 2002. How small northern gannet colonies grow faster than big ones. Waterbirds, 25: 442–448.

Munoz Cifuentes, J., Becker, P. H., Sommer, U., Pacheco, P., and Schlatter R. 2003. Seabird eggs as bioindicators of chemical contamination in Chile. Environmental Pollution, 126: 123–137.

Oro, D., Cam, E., Pradel, R., and Martínez-Abraín, A. 2004. Influence of food availability on demography and local population dynamics in a long-lived seabird. Proceedings of the Royal Society of London Biological Sciences, 271: 387–396.

Pannekoek, J., and van Strien, A. J. 2001. TRIM 3 Manual. TRends and Indices for Monitoring data. Research paper no. 0102, Statistics Netherlands, Voorburg, The Netherlands.

Pollock, K. H., Nichols, J. D., Brownie, C., and Hines, J. D. 1990. Statistical inference for capture-recpature experiments. Wildlife Monographs, 107.

Roletto, J., Mortenson, J., Harrald, I., Hall, J., and Grella, L. 2003. Beached bird surveys and chronic oil pollution in central California. Marine Ornithology, 31: 21–28.

Sandvik, H., Erikstad, K. E., Barrett, R. T., and Yoccoz, N. G. 2005. The effect of climate on adult survival in five species of North Atlantic seabirds. Journal of Animal Ecology, 74: 817–831.

Scheifler, R., Gauthier-Clerc, M., Le Bohec, C., Crini, N., Coeurdassier, M., Badot, P. M., Giraudoux, P., and Le Maho, Y. 2005. Mercury concentrations in king penguin (*Aptenodytes patagonicus*) feathers at Crozet Islands (Sub-Antarctic): Temporal trend between 1966–1974 and 2000–2001. Environmental Toxicology and Chemistry, 24: 125–128.

Seys, J., Offringa, H., Meire, P., van Waeyenberge, J., and Kuijken, E. 2002. Long-term changes in oil pollution off the Belgian coast: Evidence from beached bird monitoring. Belgian Journal of Zoology, 132: 111–118.

Seys, J., Offringa, H., van Waeyenberge, J., Meire, P., and Kuijken, E. 2002. An evaluation of beached bird monitoring approaches. Marine Pollution Bulletin, 44: 322–333.

Sims, M., Wanless, S., Harris, M. P., Mitchell, P. I., and Elston, D. A. In press. Evaluating the power of monitoring plot designs for detecting long-term trends in the numbers of common guillemots. Journal of Applied Ecology.

Spear, L. B., Ainley, D. G., and Ribic, C. A. 1995. Incidence of plastic in seabirds from the tropical Pacific, 1984–91: relation with distribution of species, sex, age, season, year and body weight. Marine Environmental Research, 40: 123–146.

Steidl, R. J., and Thomas, L. 2001. Power analysis and experimental design. Pp. 14–36 *in* Schneider, S. M., and Gurevitch, J. (eds). *Design and analysis of ecological experiments*. Oxford, UK.

Tavares P. C., Monteiro, L. R., Lopes, R. J., Pereira, M. E., Duarte, A. C., and Furness R. W. 2005. Variation in mercury contamination of little tern *Sterna albifrons* in south-west Europe: Brood, age, and colony-related effects. Bulletin of Environmental Contamination and Toxicology, 74: 177–183.

Ter Braak, C. J.F., van Strien, A. J., Meijer, R., and Verstrael, T. J. 1994. Analysis of monitoring data for many missing values: which method? *In: Bird Numbers 1992: Distribution, monitoring and ecological aspects*. Pp. 663–673. Ed by Hagemeijer, W., and Verstrael, T. J. SOVON, Beek-Ubbergen, The Netherlands.

Thyen, S., Becker, P. H., and Behmann, H. 2000. Organochlorine and mercury contamination of little terns (*Sterna albifrons*) breeding at the western Baltic Sea, 1978–96. Environmental Pollution, 108: 225–238.

van Franeker, J. A., Heubeck, M., Fairclough, K., Turner, D. M., Grantham, M. *et al.* 2005. Save the 'North Sea' Fulmar study 2002–2004: a regional pilot project for the Fulmar-Litter EcoQO in the OSPAR area. Alterra-Rapport, 1162: 1–70.

Verreault, J., Letcher, R. J., Muir, D. C.G., Chu, S., Gebbink, W. A., and Gabrielsen, G. W. 2005. New organochlorine contaminants and metabolites in plasma and eggs of glaucous gulls (*Larus hyperboreus*) from the Norwegian Arctic. Environmental Toxicology and Chemistry, 24: 2486–2499.

Votier, S. C., Hatchwell, B. J., Beckerman, A., McCleery, R. H., Hunter, F. M., Pellatt, J., Trinder, M., and Birkhead, T. R. 2005. Oil pollution and climate have wide-scale impacts on seabirds demographics. Ecology Letters, 8, 1157–1164.

Walsh, P. M., Halley, D. J., Harris, M. P., del Nevo, A., Sim, I. M.W., and Tasker, M. L. 1995. *Seabird monitoring handbook for Britain and Ireland*. NCC/RSPB/ITE/Seabird Group, Peterborough.

White, G. C., and Burnham, K. P. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study, 46: 120–139.

Williams, B. K., Nichols, J. D., and Conroy, M. J. 2002 Analysis and management of animal populations. Academic Press, USA.

Wooller, R. D., Bradley, J. S., and Croxall, J. P. 1992. Long-term Population Studies of Seabirds. Trends in Ecology and Evolution, 7: 111–114.

Yoccoz, N. G., Nichols, J. D., and Boulinier, T. 2001. Monitoring biological diversity in space and time. Trends in Ecology and Evolution, 16: 446–453.

# 5    Methods of and recommendations for sampling diet and for reporting results of dietary studies in seabirds

The development of recommendations on how to sample seabird diet and report results in a standard manner is dependent on a thorough review of the methods being used today. This was started at the WGSE 2006 meeting, and the completion deferred to 2007.

## 5.1    Introduction

Many methods have been, and still are being used to study the diet of seabirds. Some are based on opportunism whereby samples are collected *ad hoc* (e.g. from watching food uptake directly or collecting dropped fish, regurgitated food or pellets, or faeces). Others take a more systematic approach through regular collections or sightings made during a given time window. Techniques vary greatly and range from the direct killing of birds to inspect their stomach contents through to totally non-disturbing and repeatable observations of fish-carrying birds, or the indirect methods including observations of feeding flocks, analyses of faeces, regurgitated food remains (including pellets), and tissue collection for stable isotope or fatty-acid analyses. All methods have biases of one kind or other (Duffy and Jackson, 1986; Rodway and Montevecchi, 1996; Carss, 1997; González-Solís *et al.*, 1997; Andersen *et al.*, 2004), and the vast majority are restricted to the short breeding season when birds are readily accessible on or near land.

As to the majority of the year when seabirds are spread along the coasts and over the open seas, there is no completely satisfactory non-destructive method for sampling their diet. As a result, far too little is known about what and how much seabirds eat when they are at sea, outside the breeding season or for immature and non-breeding birds.

### 5.1.1    Differences in food between adults and chicks, breeders and non-breeders

When analysing and evaluating studies on seabird food, one has to be aware of possible bias that results from a non-representative sampling design. Because it is extremely difficult to sample diet of seabirds at sea, it is not surprising that the vast majority of studies on seabird feeding ecology are restricted to the breeding sites and breeding times. Furthermore, even the comparatively few studies that have compared the diets of adults vs. chicks or of breeders vs. non-breeders have almost exclusively revealed substantial differences in diet:

Seabirds provisioning food to chicks face different constraints than when self-feeding and as a result, chick food normally differs from the food taken by adults (Ydenberg, 1994). Small chicks may physically be unable to ingest large prey (e.g. Shealer, 1998); parents flying with prey visible in their bills may be subject to kleptoparasitism (e.g. Veen, 1977; Furness, 1978; Burger and Gochfeld, 1991; Ratcliffe *et al.,* 1997) or face aerodynamic or gravity constraints. Moreover, optimal prey for adults may be available only at distances from colonies that are too large for commuting (e.g. Weimerskirch, 1998). These constraints all lead to a shift away from prey optimal for chick rearing and hence accessible for diet studies in the breeding colonies. Optimal foraging theory or more precisely, central-place foraging theories (Orians and Pearson, 1979) predict that:

1 ) single prey loaders (such as guillemots or terns) should bring larger, and in energetical terms richer, prey to their chicks than they swallow themselves (Wilson *et al.*, 2004; Sonntag and Hüppop, 2005);

2 ) multiple prey loaders (such as many smaller auks that can carry several fish in their bill, Procellariiformes that convert prey to stomach oil, or seabirds that ferry multiple prey in their crop and/or stomach) should optimise their energy load per

trip, particularly if trips are long, or few and far between (Ydenberg, 1994; Davoren and Burger, 1999).

Optimising energy load may be achieved by selecting fatty fish (such as clupeids, sandeels, capelin, mackerel) but also by selecting larger fish, as these generally contain more energy per item and per gram (Hislop *et al.,* 1991; Lawson *et al.*, 1998). Birds that need only to feed for themselves may satisfy their daily needs with small or lean prey, if these are easily available, but parents that need to invest heavily in prey transport will benefit from being selective (Mehlum, 2001). Parents also need to sustain themselves and should thus attempt to optimally allocate their resources between themselves and their chicks. Optimal prey allocation may lead to letting the young starve if adult survival or fitness is at risk in poor food years. Seabirds generally are long-lived birds that will rather desert their offspring when conditions turn bad, than putting their own survival and thus their further lifetime reproductive potential at risk (e.g. Erikstad *et al.,* 1998). Thus, they only invest in young (engage in provisioning) when resources are adequate. When single-prey loaders feed their young are being fed, the allocation of food between the parents and the chicks could, in theory, take the form of optimal sharing (Leopold *et al.*, 1996; Sonntag and Hüppop, 2005), i.e. the parents ingest all small prey, while flying off only with large prey, with the threshold being determined by their relative needs. Alternatively, parents could fulfil their own needs first, before switching to provisioning. Studies that simultaneously have looked at adult and chick diet are rare in seabirds (Brown and Ewins, 1996; Davoren and Burger, 1999; Dierschke and Hüppop, 2003).

Breeding birds without chicks could either be birds that still have eggs or birds that have lost their clutch or brood. Studies generally showed that birds with chicks bring in food that is higher in energetic density than the food taken by birds without chicks (Keijl *et al.*, 1986; Noordhuis and Spaans, 1992; Brown and Ewins, 1996). Mehlum (2001) showed that common guillemots and Brünnich's guillemots that bring fish to their young can have much smaller prey, euphausiids, as their staple diet when self-feeding. This has also been shown to be the case in the pre-fledging period of common guillemot shortly after the chick has left the breeding shelf and is being fed at sea by the male parent (Anker-Nilssen and Nygård, 1987). Other studies on the diets of seabirds in the non-breeding season, i.e. away from the colonies and not connected to provisioning suggest that seabirds then take a larger variety of prey, including many species that are relatively low in energy density (e.g. Bradstreet and Brown, 1985; Hedd and Montevecchi, 2006).

Also the diet of adults may alter through the breeding period or may differ between sexes, reflecting changing demands e.g. for egg-production (Spaans, 1971; Pierotti and Annett, 1987, 1991; Pons, 1994), or even between individuals. For example, Niebuhr (1983) observed that female herring gulls in the pre-laying period preferred mussels, which provide calcium for egg-shell formation, whereas males fed on refuse. Despite the higher energetic value of refuse, mussel specialists produce more offspring, being larger at all developmental stages compared to refuse specialists (Pierotti and Annett, 1987). Things may additionally be complicated by individual feeding preferences (McCleery and Sibly, 1986) making large sample sizes necessary.

In seabird colonies, there are generally a high proportion of non-breeders present (e.g. Aebischer, 1986; Pons and Migot, 1995; Warham, 1996; Grunsky-Schönberg, 1998). These are birds that skip breeding for a year (or more), which might extenuate energetic constraints and hence increase lifetime reproduction and overall fitness (Calladine and Harris, 1997; Cam *et al.*, 1998; Bradley *et al.*, 2000). Again, due to different demands and constraints, their diets likely differ from those of breeders.

The major caveat in seabird diet studies is knowledge concerning the food of birds at sea away from the colonies, i.e. outside the breeding season, and for non-breeding and immature birds. Most species of seabird spend the majority of their lives offshore, and most data on the diet of

non-breeding seabirds is available from beached birds or from birds drowned in fishery nets. In general, feeding can be more opportunistic outside the breeding season since birds are not forced to stay in the vicinity of their breeding sites or to provision chicks. Hence food composition is more varied outside the breeding season (e.g. Spaans, 1971; Halley *et al.*, 1995; Ainley *et al.*, 1996; Ouwehand *et al.*, 2004; Ludynia *et al.*, 2005).

## 5.2 Stomach sampling/regurgitations

In order to assess the diets of seabirds, it is necessary to extract items from the digestive tract, sort and identify them and take measurements such as mass, linear dimensions (length, height etc.) and volume. Some items extracted will be considerably damaged or even lost due to digestion, so it is often necessary to estimate mass and dimensions of the original food item based on fragments. Food items may be found in the crop, proventriculus, oesophagus, gizzard or small intestine. Generally, the only items retained in the gizzard are hard parts such as bones, shells, exoskeletons, polychaete jaws and squid beaks. Everything from the proventriculus up to the oral cavity can often be sampled by lavage without harm to the bird. Sampling the gizzard or intestine, however, is only possible from dead, dissected specimens.

Size of digested prey can often be estimated from measurements of undigested hard parts such as otoliths, bullae, bones, shells and squid beaks, but the accuracy depends greatly on the amount of digestion and wear of these items (see Section 5.4 Pellets).

### 5.2.1 Dead Birds

Shooting birds at sea is one way of obtaining dietary data that is relatively free of bias resulting from differential attraction to any kind of trap (except for ship-followers). Shooting has the obvious limitation of killing the birds, which tables the issue of ethical concerns and which can have significant consequences for long-lived and endangered species. Shooting is becoming increasingly unacceptable as a tool for sampling diet. However, birds shot for other reasons, e.g. for pollutant analyses, harvesting (such as the Newfoundland guillemot hunt) or shot as pests (although those killed at e.g. aquaculture sites may provide very biased data), etc. have been used for diet studies (e.g. Rowe *et al.*, 2000). Other sources of dead birds are oiling incidents, bycatches in fishing gear and beached carcasses of oiled or wrecked birds (e.g. Blake 1983, Lorentsen and Anker-Nilssen, 1999; Ouwehand *et al.*, 2004), although beached birds have often starved to death and yield few or biased data. Dead birds may arrive on beaches in a trickle, e.g. as a result of chronic oil pollution, or may hit a coastline in masses, e.g. after a major oiling incident or wreck following extended extreme wind conditions (e.g. Stenhouse and Montevecchi, 1996). Such large-scale events should be seized for diet studies whenever possible, as they often provide large samples across a range of species from the same time and location (see e.g. Ouwehand *et al.*, 2004), even though the logistics of such "sudden samples" are often difficult to deal with. In some oil spills, specimens are often sequestered for litigation purposes so may be impossible to access for years after the incident. Efforts should, however, be made to access them after litigation, as they are often then discarded.

The entire digestive track is usually removed from the bird as soon as possible after death and frozen or preserved in either ethanol or formaldehyde. A substantial fraction (often 30% or more) of birds shot at sea do not, however, contain any food items other than bony fragments in the gizzard. Therefore substantial numbers need to be shot in order to obtain a sample large enough to ensure representation of all prey items taken.

Care needs be taken for the differential digestion of food items in different portions of the digestive track. Items in the crop can be near intact, but the further an item progresses through the digestive track, the more it is digested and consequently the more difficult it may be to identify. Items in the gizzard may be retained for a long time; sometimes until they are

forcibly regurgitated as a pellet. Squid beaks or polychaete jaws, for example, may be retained for a month or longer, and this retention needs to be taken into account when estimating dietary composition based on dissected dead birds and/or regurgitated samples. As the soft parts of squids or polychaetes are digested quickly, the beaks or jaws in the gizzard are often the only evidence of their importance in the diet. Nevertheless, using the number of these items in the gizzard will likely overestimate their proportional contribution due to long retention times.

### 5.2.2    Regurgitations

Some birds, especially nocturnal petrels, when attracted to lights at night become disoriented and land upon a ship's deck or the ground. To lessen mass or as a panic response, they vomit the contents of the upper intestinal tract. At breeding colonies, storm-petrels can also be caught in mist-nests where they will regurgitate or be induced to regurgitate (Montevecchi *et al.*, 1992; Hedd and Montevecchi, 2006). Sampling this way can be especially valuable as it may be the only way to obtain dietary information from birds at sea and/or outside the breeding season. The problem with this technique of sampling outside the breeding season is that it is entirely opportunistic and dependent on certain weather conditions, as birds are much more likely to be attracted to lights during foggy, overcast and/or rainy weather. Nevertheless, such sampling can produce valuable information on the food types available at prey patches at sea.

Other species like gannets, cormorants, gulls and terns on the nest or on their way to feed chicks may regurgitate food held in the proventriculus if disturbed. Chicks also spontaneously regurgitate in response to disturbance, or can be easily stimulated to regurgitate. Such samples may often be of little or only partly digested material which is readily identifiable in the field (e.g. gannets, cormorants) or on return to the laboratory (gulls, kittiwakes) such that the data may be fairly useful for estimates of dietary diversity. Another advantage is that it can be repeated (using different birds each time) throughout the breeding season and even on the same known individuals in some circumstances. The hard body parts (otoliths, bones, etc.) are also often not worn by digestion (although there is a differentiation in digestion rates between opaque and hyaline otoliths) thus allowing reliable determinations of prey size and hence energy content. Note, however, that the proportion of ingested items in the regurgitations is variable, so one cannot use the amount regurgitated as an estimate of total crop contents or meal size. Another limitation of this method is that the disturbance involved in some breeding colonies reduces the numbers of visits possible.

### 5.2.3    Stomach lavage, emetics

If a bird does not regurgitate "voluntarily", the upper intestinal tract can be sampled without harming the bird by flushing the contents out with water. This process, referred to as lavage, stomach flushing or water off-loading, involves pumping salt water through a tube inserted in the oesophagus of a bird and catching the regurgitated contents in a bag, sieve or bucket (Wilson, 1984; Ryan and Jackson, 1986). A latex tube is inserted deep into the bird's oesophagus, and salt water pumped (using a syringe) in the other end of the tube. The bird is then inverted over a suitable receptacle into which the water and stomach contents are emptied. The process should be repeated to ensure as complete an emptying of the gastric system as possible.

The limitations of lavage relate to how the birds are captured in the first place; as many birds vomit immediately upon being captured in a mist net or trap, so appear to be empty upon having their stomachs flushed. It has also proved difficult to use in some groups of seabird that do not regurgitate food to offspring, e.g. auks, but see Wilson *et al*. (2004).

Birds do not always eject all contents of the upper gut tract during lavage, and can be induced to do so using an emetic (Ryan and Jackson, 1986) such as the Texel ferry coffee (K. Camphuysen, pers. comm.).

## 5.3 Faeces

Bird faeces have been used in various ways to reconstruct diets. Hard parts from prey, such as bones, scales, eggs or otoliths of fish, parts of the exoskeletons of crustaceans, jaws of squid and nereid worms, setae of nereid worms, calcite plates and spines of echinoderms or shell hinges of molluscs may all survive digestions and are often excreted with the faeces. If such parts can be identified and still bear a relationship with original prey size, these may be used to identify prey and reconstruct prey size. This method has been applied to many different piscivores, most notably pinnipeds and otters (e.g. Pierce *et al.*, 1991; Kingston *et al.,* 1999; Andersen *et al.*, 2004; Tollit *et al.,* 2004). Seabirds that excrete such remains through their faeces are also candidates for similar studies and many have been carried out on omnivorous gulls and skuas (e.g. Andersson and Götmark, 1980; Ambrose, 1986; Kubetzki *et al.*, 1999; Kubetzki and Garthe, 2003), piscivorous ducks (Rodway and Cooke, 2002), mollusc-eating seaduck (Swennen, 1976; Nehls, 1989; Nehls and Ketzenberg, 2002; Leopold *et al,*. submitted), benthos-feeding waders (e.g. Dekinga and Piersma, 1993; Scheiffarth, 2001) and other birds (e.g. Ormerod and Tyler, 1991; Taylor and O'Halloran, 1997). Relatively few studies have, however, been carried out on some species (e.g. terns Veen *et al.,* 2003).

Advantages of the method are that it is non-invasive and low-tech. Furthermore, large sample sizes can be processed and time series can be built by repeated sampling schemes. Given that different methods often reveal different prey types, studying remains in faeces may uncover prey species previously unknown, e.g. *Nereis* jaws in sandwich tern faeces (Stienen *et al.*, unpubl. data). Faecal or scat samples can be used to identify the sex or even individual of the predator involved, allowing sex-specific studies of diet (Reed *et al.*, 1997). Being widely used and with samples being readily available, particularly in seals, the method has been rather extensively tested against other diet study methods (Prime and Hammond, 1987; Dellinger and Trillmich 1988; Cottrell *et al.,* 1996).

These tests have, however, demonstrated that studies of faeces, as are methods covered in 5.2, are unlikely to reveal all prey taken by the predator. Some prey do not survive digestion in a way that would allow finding traces in the faeces, while some birds also use other means to rid themselves of prey hard parts, e.g. through regurgitation of pellets (see below). Faeces are unlikely to be collected offshore, at sea, unless a suitable platform on which faeces are deposited are available for sampling (e.g. Camphuysen and de Vreeze, 2005). Processing faecal samples can be unpleasant although several washing methods have been developed (Bigg and Olesiuk, 1990; Brasseur and Janssen, in prep.). It is also time-consuming compared to e.g. measuring whole fish in a bird's oesophagus, and good reference collections (e.g. Härkönen, 1986; Watt *et al.*, 1997; Leopold *et al.*, 2001) are required. Prey remains are bound to be worn after passage through the predator's gut, and correction for wear and tear is needed. Some parts survive better than others and some prey may be completely overlooked or greatly underestimated.

## 5.4 Pellets

Several seabirds eject indigestible prey remains in regurgitated pellets. These may be collected and the remains sorted out, using similar methods to those described under "Faeces". This method has been widely used on cormorants and shags (e.g. Kennedy and Greer, 1988; Barrett *et al.*, 1990; Hald-Mortensen, 1995; Grémillet and Argentin, 1998; Leopold *et al.,* 1998; Olmos *et al.*, 2000), pelicans (e.g. Derby and Lovvorn, 1997), gulls (Meijering, 1954; Spaans, 1971; Wietfeld, 1977; Kubetzki *et al.*, 1999; Kubetzki and Garthe, 2003), terns (e.g.

Granadeiro *et al.*, 2002; Veen *et al.,* 2003) and other birds such as waders, kingfishers, dippers (Swennen, 1971; Jost, 1975; Cairns, 1998). Being widely used and with samples being readily available, particularly in cormorants, the method has been extensively tested against other diet study methods (Brugger, 1993; Harris and Wanless, 1993; Russell *et al.*, 1995; Trauttmansdorff and Wassermann, 1995; Zijlstra and van Eerden, 1995; Suter and Morel, 1996; Casaux *et al.,* 1997, 1999).

A major advantage of this method is that it is non-invasive and low-tech. Large sample sizes can be processed and time series can be built by repeated sampling schemes. The method is also quantitative, on the assumption that birds generally eject one pellet per day and that this pellet will contain the hard parts of all prey eaten. Although these assumptions are often violated, pellet studies do allow some quantification of diet. In very few cases, the amount of pellets produced has been verified in the field, e.g. in cormorants (Hüppop and Fründt, 2002). The method is, however, better for determination of diet composition rather than for quantification of consumption.

Many different prey types have been found in pellets, including unexpected ones (e.g. Leopold and van Damme, 2003), suggesting that indeed most (but not all – see below) prey can be assessed in the pellets. Although finding pellets is often restricted to breeding colonies or roosts, this is usually not a great problem as the birds concerned are largely feeding locally and inshore. Pellets can be collected from any dry surface where the target birds breed or roost, such as offshore lighthouses and platforms, or even especially designed floating pellet-collecting devices (Gagliardi *et al.*, 2003).

Some comparative studies have, however, clearly indicated that particularly the hard parts of small prey, may not end up in the pellets but rather in faeces (e.g. Veen *et al.,* 2003). As in faecal studies, some prey do not leave hard parts in pellets and processing pellets and making reconstructions of numbers of prey and prey sizes is time consuming.

Further problems arise as a result of the possibility of secondary consumption of prey by the seabird, i.e. the pellet may contain remains of prey present in the digestive tract of the fish consumed by the seabird. For example, Johnson *et al.* (1997) suggested that the invertebrate prey found in the pellets of double-crested cormorants were prey of the fish consumed and not of the cormorants themselves. This source of error may also be relevant in faecal studies, those of regurgitated remains and in analyses of dead birds containing partly or completely digested material.

## 5.5   Archaeological: guano, middens and mummies

Parts of pellets or faeces can be conserved in sediments (Martini and Reichenbacher, 1993). Hence, geological deposits, including guano layers in recent and abandoned seabird colonies (e.g. Rand, 1960) or archaeological sites may contain information on the diets of seabirds in the past. Perhaps even more spectacular, although of little relevance to modern diet studies of seabird, prey remains are sometimes found in fossil seabirds (e.g. Mayr, 2004).

## 5.6   *Food dropped in the colony*

Food, often fish, dropped by adults returning to the colony or dropped by chicks during feeding may be found on the ground or on breeding ledges, where they are readily collected and identified. They are, however, poor indicators of food choice. In mixed colonies, the species that dropped the food is generally unknown, and those dropped by chicks may be unrepresentative of the fish normally eaten. For example, guillemot and tern chicks often reject fish that are too large or too difficult to swallow, or those dropped by displaying guillemots (often non-breeding birds) may not be representative of those caught by chick-

feeding adults. A further source of bias is the fact that dropped fish are often readily found and eaten by other seabirds in the area, with the most conspicuous items disappearing first.

## 5.7    Observations and collection of food from fish-carrying species

Some seabird species bring whole fish (and sometimes but rarely other food items) carried openly cross- or lengthwise in the bill to their chicks. Some seaducks bring large prey items to the surface before swallowing them. With practice and for species carrying single or few fish, e.g. terns, guillemots, razorbills, black guillemots and puffins, it is generally easy to identify such fish from a distance using binoculars or telescope as the bird stands in the colony (Birkhead and Nettleship, 1987; Harris and Wanless, 1995; Rodway and Montevecchi, 1996; Larson and Craig, 2006). It is further possible to estimate fish size (e.g. small, medium, large) in relation to e.g. bill length. In some cases, identification and estimates of fish size can be controlled by subsequently catching the observed birds and collecting the fish (see below). For single prey loaders that carry fish lengthwise in the bill, such as guillemots, it needs to be taken into account that the head of the fish may be carried at different depths in the bird's gullet thus reducing the observed length of fish in the bill. For species carrying many small fish (e.g. puffins), species identification and quantification is also possible (and often used) but the possibilities of observation error are larger (Rodway and Montevecchi, 1996).

The main advantage of being able to make direct observations of fish is the possibility of collecting large samples without any disturbance to the birds. If the species breeds in dense colonies, e.g. guillemots, the simplicity of the method and the possibility to make many observations can be made over short time periods and also enable the documentation of short-term temporal and spatial (within or between colonies) variations in prey choice.

The main disadvantage is the possibility of misidentifying the prey with no possibility of later confirmation (unless the fish are photographed or filmed). This is even more of a problem for species carrying many small (even larval) fish, as numbers and sizes are very easily misjudged. As a result, it is often preferable to sample the fish directly by capturing the fish-carrying birds (see Rodway and Montevecchi, 1996).

In large colonies, fish-carrying common (and less often Brünnich's) guillemots on their way to the nest site can be caught easily using a fleyg net, or with a noose pole once they have landed on or near the site (Birkhead and Nettleship, 1987; Davoren and Montevecchi, 2003). Fish-carrying puffins (and razorbills) can be caught with fleyg nets as they arrive at or circle past the colony, with a noose pole once they have landed or in mist nets (Wanless *et al.*, 2004). Because small fish, fry or larvae are easily lost in the undergrowth (or even over the cliff edge!), sampling sites should be chosen with care. Trimming the vegetation or placing plastic sheeting under the mist net reduces the problem of fish lost in vegetation. A second method is to photograph, videofilm or "digiscope" (digital photography using binoculars or telescope) fish being carried in the bill (Larson and Craig, 2006). Both voucher methods enable subsequent confirmation of identification by fish experts and possibly more accurate estimates of prey size.

For burrow nesting species, a second method is to block the entrances of 20–30 burrows for 1–2 hours using a screen (of wire or plastic netting) a short distance inside or fish net placed over the entrance (Sanger and Hatch, 1987; Finney *et al.,* 2001; Montevecchi unpubl. data) and to collect any dropped fish. One problem with this method is that the samples are sometimes damaged as the adult tries to get past the screen. Successful trials have also been carried out in which the chick's bill was sealed shut using a pipe-cleaner such that it can not pick up food dropped by the adult (Harding *et al.*, 2002). These methods involve some disturbance of the adults, but the collection of fish has the great advantage of allowing accurate quantitative studies of prey composition (either by number, mass or energy content)

in that the fish are whole and (often very) fresh (even at times live) when brought into the colony.

Either method (observation or collection) is, however, limited to the chick-rearing period which, for guillemots and razorbill colonies may last only 4–5 weeks between the hatching of the first egg and the fledging of the last chick. For terns and some auks, however, the method may also be used to determine food choice and quality during the courtship period early in the breeding season), but how representative the fish fed to mates are of the general diet of the species is largely unknown. As discussed in 5.1.1, the prey spectrum brought to the colony for display or for feeding young might also differ from the diet of the adults themselves.

## 5.8   Indirect biochemical assays

Indirect methods of determining seabird diets using biochemical assays have several advantages over more traditional, direct methods. Direct diet sampling most often indicates what the individual seabird has just eaten and therefore may not reflect "average" or typical diet if temporal variability is high. In contrast, both stable isotope ratios and fatty acid signatures integrate diet information over space and time (ranging from days to months) (see Hobson *et al*., 1994). Biases associated with direct diet sampling can sometimes be large due to, for example, digestion of soft parts and preferential retention of hard parts (see above). These indirect methods do not suffer from the same problems. A major value of using chemical assays is that they provide a means to sample adult diets directly during breeding and importantly offer a means of assessing trophic interactions during non-breeding periods (e.g. Hedd and Montevecchi, 2006).

### 5.8.1   Stable isotope analysis

Examining tissue levels of different stable isotopes has been used extensively in avian feeding ecology studies over the past decade or more (e.g. Hobson and Welch, 1992; Hobson, 1993; Hobson *et al*., 1994; Sydeman *et al*., 1997). The method takes advantage of the fact that stable isotope ratios of nitrogen ($^{15}N/^{14}N$) and carbon ($^{13}C/^{12}C$) in tissues pass from prey to predator in a predictable manner. In the case of N, and to a lesser extent C, the ratio of the heavier (and rarer) isotope to the lighter (and more common) one increases at a rate of about 3–5 parts per thousand between each trophic level in marine systems. Therefore the method indicates trophic level of the predator, not the specific items in the diet (unless the diet is very simple). Although complex, stable isotope methodologies are now fairly routine and laboratories around the world offer this service at a reasonable cost.

Because the metabolic rates of various tissues differ, stable isotope ratios reflect trophic level at different time (and hence spatial) scales from days in the case of "fast" tissues (e.g. blood) to months in the case of "slow" ones (e.g. muscle, feathers) (Hobson *et al*., 1994; Bearhop *et al*., 1999). The method does, however, require voucher samples from hypothesised foraging areas. Stable isotope analysis of "slow" tissues provides the opportunity to assess diet during times of the year not normally covered with traditional diet sampling at seabird breeding colonies. Also by analyzing small pieces of feathers grown in the non-breeding season, assessments of fall, winter and spring diets can be possible depending on the species' moulting chronology (e.g. Hedd and Montevecchi, 2006). Some tissues (e.g. bone collagen) that can be collected from specimens of extinct birds or from bird remains collected at archaeological sites can be used to assess the trophic interactions of extinct and ancient birds (e.g. Hobson and Montevecchi 1993).

Although carbon isotope ratios change less between trophic levels than N, they are useful in providing a general idea of how far from shore the bird feeds or in which oceanographic regions. Carbon-13 is enriched in relation to $^{12}C$ in nearshore compared to offshore waters and from high latitudes toward the equator (Rau *et al*., 1982; Cherel *et al*,. 2000).

### 5.8.2 Quantitative fatty acid signature analysis

A relatively new method to probe the diets of marine organisms takes advantage of the fact that 1) the fatty acid composition of prey species is diverse (between species) and characteristic (within species), 2) long-chain (i.e. >14 units) fatty acids pass relatively un-degraded to predators, and 3) the predator ultimately stores prey fatty acids in adipose tissue, which can be non-destructively sampled using biopsy (Iverson *et al*., 2004). As relatively few fatty acids are synthesised by the predators themselves, dietary versus intrinsic fatty acids can be distinguished. This technique has advantages over using stable isotopes because actual diet composition rather than just trophic level can be assessed (Käkelä *et al.,* 2006, 2007). The potential of using the two techniques in combination offers the possibility for more robust chemical analyses.

A problem inherent in the technique is that predator diets usually contain more than one prey species such that the fatty acid signatures are often complex and cannot be examined just by eye. Furthermore, variability of fatty acid signatures between individuals of prey species, and intrinsic predator fatty acid production and metabolism sometimes need to be taken into account when interpreting the predator signatures. Iverson *et al*. (2004) outline a statistical modelling technique that was successful in estimating known diet composition of marine seals and mink. They suggest that the technique has wide application for other marine predators such as seabirds, and more recently, Iverson and Springer (in prep.) have confirmed the applicability of the technique to seabirds breeding in Alaska.

The technique is demanding because a fatty acid database of all possible prey is needed to accurately interpret predator signatures. The database for seabird diets in the Atlantic will no doubt expand over the next few years (and is already doing so in Alaska, see Iverson and Springer in prep). It is also important to calibrate the metabolic shifts in fatty acid signatures between the consumer seabird and its food (Käkelä *et al.,* 2005). The availability of software to perform the statistical modelling requirement of the method would aid its general applicability.

### 5.8.3 Serological methods

Serological methods also have the potential for detecting species-specific markers in digested prey items. The enzyme-linked immunosorbent assay (ELISA) has been used for identification of invertebrate tissue but it requires considerable laboratory effort to produce specific antisera to the range of potential prey species (Freeman and Smith, 1998). Trials to identify fish and molluscan prey of jackass penguins also noted problems with cross reactivity (Walter *et al*., 1986).

Pierce *et al*. (1990) tested the application of serological methods to the identification of fish prey in the diets of marine mammals. Antisera were raised to muscle protein extracts of three fish species. The antisera were tested for reactions with protein extracts from raw and *in vitro* digested fish muscle, stomach contents of captive bottlenose dolphins fed on known diets, digestive tract contents of grey and common seals (that contained hard remains of known prey species), and faeces of captive seals fed on known diets. The salmon antisera were shown to be sufficiently strong and specific to be used for identification of salmonid proteins in digestive tract contents of marine mammals, and were potentially applicable to screening seal faeces. Antisera raised for cod and herring were less successful, due to low specificity and low titre, respectively.

Due to the high number of prey species in most seabirds and the need for a reference database, serological methods presumably have a too high cost-effect ratio in most cases.

### 5.8.4    Gel electrophoresis and iso-electric focusing of proteins

Walter and O'Neill (1986) tested polyacrylamide gel electrophoresis to identify prey consumed by jackass penguins. They found that different prey species could be recognised up to 6 h after ingestion. Freeman and West (1998) used iso-electric focusing to identify fish tissue in Westland petrel diet samples. Forty-five percent of the samples from Westland petrel stomachs produced clear protein banding patterns and more than half of these were identified as species common in fisheries' waste. Proteins in the other samples were presumably too digested for this technique. Despite Freeman and West (1998) claim that iso-electric focusing is a comparatively quick and inexpensive technique and is particularly useful for diet studies where flesh eaten is likely to be relatively undigested at the time of sampling and despite the fact that the method is widely used in fisheries studies, the method has not been used in seabird diet studies, nor has gel electrophoresis.

## 5.9    Food sampling under feeding birds

### 5.9.1    Fish/plankton hauls under seabird feeding frenzies

When flocks of intensively feeding seabirds are encountered, sampling the sea for potential food items will provide direct information of the potential prey locally available. Such sampling can be done while conducting direct observations on the feeding birds, or sampling these birds in any other way. Food may be sampled by taking fish or plankton hauls at the site, or acoustically. Fish hauls are often taken opportunistically, that is only when feeding frenzies of seabirds are encountered. It would be useful to also sample blanks, i.e. at similar locations away from the feeding frenzies. Both seabirds feeding on schools of fish and schools of plankton are eligible for this approach. Examples can be found in Grover and Olla (1983), Skov *et al.* (1989), Piatt (1990), Baars *et al.* (1990), Camphuysen (1999) and Frengen and Thingstad (2002).

### 5.9.2    Benthos sampling under flocks of seaduck

Flocks of seaducks that reside for a longer period at a certain location are likely to feed there, on benthic prey. Because benthic prey tends to stay the same place (possible exceptions being fish eggs, amphipods and other epi-benthos), such locations may be sampled with bottom grabs, dredges, nets, cameras on remotely operated vehicles or other devices to assess available potential prey. In situations where one prey type is clearly dominantly present, and suitable as food, it may be inferred that this potential prey is also the actual prey taken by the ducks. Examples are given in Leopold *et al.* (1995), Kube (1996) and Degraer *et al.* (1999).

## 5.10  Application of data loggers

Data loggers have been applied successfully to study the timing of feeding and the amount of food ingested. Generally, all birds need to be caught first and the devices have to be deployed. Secondly, the birds need to be recaptured to download the data and/or remove the devices. So far, stomach temperature loggers have been most commonly applied. Their use is based on the principle that the ingestion of cold prey (fish, cephalopods etc.) by the warm-blooded seabirds leads to a drop in temperature (Wilson *et al.*, 1992). From the magnitude of the temperature drop and the time it takes to re-warm the stomach and contents, the amount of food can be calculated (e.g. Wilson *et al.*, 1995). This method has been successfully applied to a variety of seabirds including penguins, albatrosses, cormorants and gannets (e.g. Grémillet and Plös, 1994; Wilson *et al.* 1995; Garthe *et al.*, 1999). A major problem with this technique is that the detection works very well for single, large prey items but less well for multiple prey items, especially small ones. In the worst case, many small fish such as sandeels or small clupeids cannot be detected at all after the stomach has partly filled so that both information on timing of feeding and amount of food could be masked (Wilson *et al.*, 1995; Wanless *et al.*, 2005).

However, some studies were able to quantify prey consumption. In order to avoid the masking effect of prey lying in the stomach on top of the device, two other technological developments were developed that try to detect prey ingestion in the bird before the prey enters the stomach. Ancel *et al*. (1997), Charassin *et al*. (2000) and others applied sensors in the oesophagus that record prey ingestion while the prey moves from the beak of the birds towards the stomach. Wilson *et al*. (2002) have recently devised a mandibular sensor that record changes in sensor voltage, proportional to magnetic field strength, and thus inter-mandibular angle. Captive feeding trials showed that prey mass could be determined with reasonable accuracy, and there was also some indication that prey type could be resolved if recording frequency were high enough (Wilson *et al*., 2002). Heart rate monitors have been used on albatrosses and White-chinned Petrels (Bevan *et al.,* 1995) on the principle that heart rate increases in response to ingestion and preliminary digestion of food.

## 5.11 Presentation of data

The large variety of data collection necessitates, besides standard methods of sampling, a unification of how to present the results. Duffy and Jackson (1986) reviewed methods for analysing and presenting dietary data, and this is still an excellent reference 20 years after publication. The main objectives of diet analyses generally are 1) to compare diet composition between species, times and sites and 2) to quantify the consumption rate of a predator on its prey on a species-level and, in fish-eating species, possibly also on a cohort-level. Hence the data have to be presented in a way to fulfil these aims and to allow inter-study comparisons.

Data collections and presentations are based on research objectives. For example, research questions may focus on food webs and prey consumption by avian predators or on chick growth and success. In the latter instance, more emphasis is placed on the sizes of food loads fed to chicks (regurgitated feed may contain many partially digested prey) and how these change over time. As well researchers may be interested in the nutrient and organic composition and the energy densities of chick feeds. When parents feed their offspring whole, relatively undigested fish, chick diets may be more directly and easily related to prey consumption assessments.

It is also essential to report the sites and times of sample collections, as prey species clearly vary widely and often irregularly over space-time dimensions. The diets of conspecific seabirds can vary considerably among colonies and oceanographic regions (e.g. Schneider and Hunt 1982; Barrett *et al*. 1987, Montevecchi *et al.,* 1992; Garthe *et al.,* 2007), as well as seasonally (e.g. Barrett *et al*. 1987, Hedd and Montevechi, 2006). Thus, some estimate of spatial and temporal variability in data presented needs to be provided, as too often single-day or single-location data are presented as being representative of a species (see Brown *et al.,* 1981). Seabirds are opportunistics and to a certain extent capture what prey are available.

### 5.11.1 Qualitative data/Taxonomy

Prey items are usually identified to the lowest achievable taxonomic level (order, family, genus, species (and sometimes subspecies)). It should be made clear in a final publication what reference works have been used or what taxonomic conventions were followed so that readers can understand what taxa are listed. In the absence of a standardised world list of animals and plants, and given the frequent changes in nomenclature following advances in taxonomic research, an author should always try and facilitate future scientists by referring to text-books which were used to identify prey. A full list of reported taxa should be added as an (electronic) appendix of each diet study, to facilitate future use, including at least the phylum and class of all prey items, and when possible also order, family, genus and (sub-)species, such as exemplified below (Table 5.1). Such a list could also include the 'common name' of prey items, as they may be have been used elsewhere in the paper.

**Table 5.1. Example of an appendix in a hypothetical diet study listing prey items found and reported, including insects, crustaceans, worms, molluscs, bony fish and plants.**

| KINGDOM | PHYLUM | CLASS | ORDER | FAMILY | GENUS | SPECIES |
|---|---|---|---|---|---|---|
| Animalia | Mandibulata | Insecta | Ephemeroptera | | | |
| Animalia | Mandibulata | Insecta | Diptera | | | |
| Animalia | Mandibulata | Insecta | Diptera | Muscidae | *Musca* | *domestica* |
| Animalia | Mandibulata | Insecta | Coleoptera | Carabidae | *Poecilus* | *versicolor* |
| Animalia | Mandibulata | Insecta | Coleoptera | Coccinellidae | | |
| Animalia | Crustacea | Malacostraca | Decapoda | Corystidae | *Corystes* | *cassivelaunus* |
| Animalia | Crustacea | Malacostraca | Decapoda | Cancridae | *Cancer* | *pagurus* |
| Animalia | Annelida | Polychaeta | Phyllodocida | Nereidae | *Nereis* | *virens* |
| Animalia | Annelida | Polychaeta | Phyllodocida | Nereidae | *Nereis* | *diversicolor* |
| Animalia | Annelida | Oligochaeta | Terricola | Lombricidae | *Lumbricus* | *terrestris* |
| Animalia | Mollusca | Bivalvia | Cardiacea | Cardiidae | *Cerastoderma* | *edule* |
| Animalia | Mollusca | Bivalvia | Mactracea | Mactridae | *Spisula* | *subtruncata* |
| Animalia | Mollusca | Bivalvia | Tellinacea | Tellinidae | *Macoma* | *balthica* |
| Animalia | Vertebrata | Osteichthyes | | | | |
| Animalia | Vertebrata | Osteichthyes | Clupeiformes | Clupeidae | *Clupea* | *harengus* |
| Animalia | Vertebrata | Osteichthyes | Clupeiformes | Clupeidae | *Sprattus* | *sprattus* |
| Animalia | Vertebrata | Osteichthyes | Gadiformes | Gadidae | *Merlangius* | *merlangus* |
| Animalia | Vertebrata | Osteichthyes | Scorpaeniformes | Triglidae | *Trigla* | *lucerna* |
| Animalia | Vertebrata | Osteichthyes | Pleuronectiformes | | | |
| Animalia | Vertebrata | Osteichthyes | Pleuronectiformes | Bothidae | *Arnoglossus* | *laterna* |
| Animalia | Vertebrata | Osteichthyes | Pleuronectiformes | Solidae | *Solea* | *solea* |
| Plantae | Angiospermae | Dicothyledones | Tubiflorae | Convolvulaceae | *Convolvus* | |
| Plantae | Embryophyta | Spermatopsida | Vitales | Vitaceae | *Vitis* | *vinifera* |
| Plantae | Magnoliophyta | | | | | |
| Plantae | Magnoliophyta | Liliopsida | Poales | Poaceae | | |
| Plantae | Magnoliophyta | Liliopsida | Poales | Poaceae | *Triticum* | |
| Plantae | Magnoliophyta | Liliopsida | Poales | Poaceae | *Ammophila* | *arenaria* |

The minimum is a list of all different taxa found. For meta-analyses it is important to regard the taxonomic level to which species were identified, namely when comparing lists from different species or sites. For example, a category "unidentified polychaetes" may comprise only a single species, a few species or even some dozen species, which makes a big difference when comparing species numbers as an indicator of e.g. biodiversity. If different taxa were identified down to different taxonomic levels (e.g. order, family, genus, species) comparisons have to take this into consideration. Also a possible bias caused by different stages of digestion for different taxa may cause severe deviations from reality. Ignoring unidentifiable diet components is likely to bias against more rapidly-digestible material (Duffy and Jackson, 1986). The same holds, of course, for quantitative analyses.

## 5.11.2 Quantitative data

Prey lists should be extended to make at least some estimations of the abundance of the different taxa found. The easiest (and fastest) way is to note in how many "sample-units" the respective food-item occurred, i.e. in how many percent of all pellets, stomachs etc., which should be termed "frequency of occurrence". However, regarding the large differences in size of prey items in many seabird species (e.g. copepods vs. fish in fulmars, Furness and Todd, 1984, or amphipods and other small crustaceans vs. fish in Brünnich's guillemot, Lønne and Gabrielsen, 1992) better measures to quantify food should also be applied. Dietary data can be quantified e.g. in terms numbers of individuals per taxa (resulting in "numerical abundance")

or by biomass estimates per taxa if there are means to count individuals and/or to estimate volume or mass, respectively. Because many prey items will be partly or largely digested and therefore incomplete, there will be a need to convert numbers of individual prey particles to the (minimum/maximum) number of individual prey of a given size and mass. Any documentation should clarify the techniques used to estimate the size and mass of individual prey (e.g. the regression equations used to calculate fish size from otoliths dimensions, or to estimate shrimp size from particular measurements of claws or carapax). Documentation of prey size should preferably include length frequency distributions (histograms) as these will often reveal the age class distributions of fish taken as well as provide the range (minimum and maximum), mean (with standard deviation or error) *and/or* median size of prey (including confidence limits) and sample size. Biomass given as wet mass is preferable to dry mass.

Several indices and methods to compare species or sites have been published. Day and Byrd (1989) developed an index of relative importance, Duffy and Jackson (1986) have listed a variety of diversity indices and Swanson *et al.* (1974) describe how to mathematically weight diet samples. Diet similarity (or overlap) among samples can be determined e.g. by using percent composition by mass and "Morisita's Index of Diet Similarity", which expresses similarity as a percent (Baltz and Morejohn, 1977). Other niche parameters that can be calculated if frequencies are available are niche breadth or niche overlap (Colwell and Futuyma, 1971; Mühlenberg, 1989). Also more sophisticated statistics such as cluster analysis or multidimensional scaling may be applied for categorizing dietary data (Lønne and Gabrielsen, 1992; Kubetzki and Garthe, 2003). All these methods require that the original data are presented in a comparable manner (see above). Again it is extremely important to have the diet analysed down to same taxonomic level to achieve comparable data and to allow for e.g. differences in digestibility.

## 5.12  References

Aebischer, N. J. 1986. Retrospective investigation of an ecological disaster in the shag, *Phalacrocorax aristotelis*: A general method based on long-term marking. Journal of Animal Ecology, 55: 613–629

Ainley, D. G., Spear, L. B., Allen, S. G., and Ribic, C. A. 1996. Temporal and spatial patterns in the diet of the common murre in California waters. Condor, 98: 691–705.

Ambrose, Jr W. G. 1986. Estimates of removal rate of *Nereis virens* (Polychaeta: Nereidae) from an intertidal mudflat by gulls (*Larus* spp.). Mar. Biol., 90: 243–247.

Ancel, A., Horning, M., and Kooyman, G. L. 1997. Prey ingestion revealed by oesophagus and stomach temperature recordings in cormorants. Journal of Experimental Biology, 200: 149–154.

Andersson, M., and Götmark, F. 1980. Social organisation and foraging ecology in the Arctic skua *Stercorarius parasiticus*: a test of the food defendability hypothesis. Oikos, 35: 63–71.

Andersen, S. M., Lydersen, C., Grahl-Nielsen, O., and Kovacs, K. M. 2004. Autumn diet of harbour seals (*Phoca vitulina*) at Prins Karls Forland, Svalbard, assessed via scat and fatty-acid analyses. Canadian Journal of Zoology, 82: 1230–1245.

Anker-Nilssen, T., and Nygård, T. 1987. Notes on the food choice of adult and young Guillemots *Uria aalge* during post-breeding migration in Central Norway. Fauna norvegica Ser. C, Cinclus, 10: 53-56.

Baars, M. A., Duineveld, G. C.A., van Duyl, F. C., de Gee A., Kraay, G. W., Leopold, M. F., Oosterhuis, S., van Raaphorst, W., and Westra, C. 1990. The Ecology of the Frisian Front. ICES C. M.1991/L:25Q.

Baltz, D. M., and Morejohn, G. V. 1977. Food habits and niche overlap of seabirds wintering on Monterey Bay, California. Auk, 94: 526–543.

Barrett, R. T., Anker-Nilssen, T., Rikardsen, F., Valde, K., Røv, N., and Vader, W. 1987. The food, growth and fledging success of Norwegian Puffin chicks *Fratercula arctica* in 1980–1983. Ornis Scandinavica, 18: 73–83.

Barrett, R. T., Røv N., Loen, J., and Montevecchi, W. A.. 1990. Diets of Shags *Phalacrocorax aristotelis* and Cormorants *P. carbo* in Norway and implications for Gadoid stock recruitment. Marine Ecology Progress Series, 66: 205–218.

Bearhop, S., Thompson, D. R., Waldron, S., Russel, I. C., Alexander, G., and Furness, R. W. 1999. Stable isotopes indicate the extent of freshwater feeding by cormorants *Phalacrocorax carbo* shot at inland fisheries in England. Journal of Applied Ecology, 36: 75–84.

Bevan, R. M., Butler, P. J., Woakes, A. J., and Prince, P. A. 1995. The energy expenditure of free-ranging Black-browed Albatrosses. Philosophical Transactions, Biological Sciences, 350: 119–131.

Bigg, M. A., and Olesiuk, P. F. 1990. An enclosed elutriator for processing marine mammal scats. Marine Mammal Science, 6: 350–355.

Birkhead, T. R., and Nettleship, D. N. 1987. Ecological relationships between common murre, *Uria aalge*, and thick-billed murre, *Uria lomvia*, at Gannet Islands, Labrador. III. Feeding ecology of the young. Canadian Journal of Zoology, 65: 1638–1649.

Blake, B. F. 1983. A comparative study of the diet of auks killed during an oil incident in the Skagerrak in January 1981. Journal of Zoology (London), 201: 1–12.

Bradley, J. S.,Wooller, R. D., and Skira, I. J. 2000. Intermittent breeding in the short-tailed shearwater *Puffinus tenuirostris*. Journal of Animal Ecology, 69: 639–650.

Bradstreet, M. S.W., and Brown, R. G.B. 1985. Feeding ecology of the Atlantic Alcidae. In: Nettleship D. N., and Birkhead, T. R. (eds). The Atlantic Alcidae. Academic Press, London/New York: pp 264–318.

Brown, K. M., and Ewins, P. J. 1996. Technique-dependent biases in determination of diet composition: an example with Ring-billed Gulls. Condor, 98 34–41.

Brown, R. G.B., Barker S. P., Gaskin, D. E., and Sandeman, M. R.. 1981. The foods of great and sooty shearwaters P*ufffinus gravis* and *P. griseus* in eastern Canadian waters. Ibis, 123: 19–30.

Brugger, K. E. 1993. Digestibility of three fish species by Double-crested Cormorants. Condor, 95: 25–32.

Burger, J., and Gochfeld, M. 1991. The Common Tern–its breeding biology and social behavior. Columbia University Press, 413 pp.

Cairns, D. K. 1998. Diet of cormorants, mergansers and kingfishers in northeastern North America. Canadian Technical Report of Fisheries and Aquatic Sciences, August 1998, I-III: 1–29.

Calladine, J., and Harris, M. P. 1997. Intermittent breeding in the Herring Gull *Larus argentatus* and the Lesser Black-backed Gull *Larus fuscus*. Ibis, 139, 259–263.

Cam, E., Hines, J. E., Monnat, J. Y., Nichols, J. D., and Danchin, E. (1998) Are adult non-breeders prudent parents? The kittiwake model. Ecology, 79, 2917–2930.

Camphuysen, C. J. 1999. Diurnal activity patterns and nocturnal group formation of wintering Common Murres in the central North Sea. Colonial Waterbirds, 21: 406–413.

Camphuysen, C. J., and Vreeze, F. de. 2005. De Drieteenmeeuw als broedvogel in Nederland. Limosa, 78: 65–74.

Carss, D. 1997. Techniques for assessing cormorant diet and food intake: towards a consensus review. Suppl. Ric. Biol. Selvaggina, XXVI: 197–230.

Casaux, R. J., Favero, M., Coria, N., and Silva, P. 1997. Diet of the imperial cormorant *Phalacrocorax atriceps*: comparison of pellets and stomach contents. Marine Ornithology, 25(1–2): 1–4.

Casaux, R. J., Favero, M., Barrera-Oro, E. R., and Silva, P. 1999. Feeding trial on an imperial cormorant *Phalacrocorax atriceps*: preliminary results on fish intake and otolith digestion. Marine Ornithology, 23: 101–106.

Charrassin, J.-B., Kato, A., Handrich, Y., Sato, K., Naito, Y., Ancel, A., Bost, C.-A., Gauthier-Clerc, M., Ropert-Coudert, Y., and Le Maho, Y. 2000. Feeding behaviour of free-ranging penguins determined by oesophageal temperature. Proceedings of the Royal Society of London, Biological Series, 268: 151–157.

Cherel, Y., Hobson, K. A., and Weimerskirch, H. 2000. Using stable isotope analysis of feathers to distinguish moulting and breeding origins of seabirds. Oecologia, 122: 155–162.

Colwell, R. K., and Futuyma, D. J. 1971. On the measurement of niche breadth and overlap. Ecology, 52: 567–576

Cottrell, P. E., Trites, A. W., and Miller, E. H. 1996. Assessing the use of hard parts in faeces to identify harbour seal prey: results of captive-feeding trials. Canadian Joirnal of Zoology, 74: 875–880.

Davoren, G. K., and Burger, A. E. 1999. Difference in prey selection and behaviour during self-feeding and chick provisioning in rhinoceros auklets. Animal Ecology, 58: 853–863.

Davoren, G. K., and Montevecchi, W. A. 2003. Signals from seabirds indicate changing biology of capelin biology. Marine Ecology Progress Series, 258: 253–261.

Day, R. H., and Byrd, G. V. 1989. Food habits of the whiskered auklet at Buldir Island, Alaska. Condor, 91: 65–72.

Degraer, S., Vincx, M., Meire, P., and Offringa, H. 1999. The macrozoobenthos of an important wintering area of the common scoter *Melanitta nigra*. Journal of the Marine Biological Association, UK, 79: 243–251.

Dekinga, A., and Piersma, T. 1993. Reconstructing diet composition on the basis of faeces in a mollusc-eating wader, the knot *Calidris canutus*. Bird Study, 40: 144–156.

Dellinger, T., and Trillmich, F. 1988. Estimating diet composition from scat analysis in otariid seals (Otariidae): is it reliable? Canadian Journal of Zoology, 66: 1865–1870.

Derby, C. E., and Lovvorn, J. R. 1997. Comparison of pellets versus collected birds for sampling diets of double-crested cormorants. Condor, 99: 549–553.

Duffy, D., and Jackson, S. 1986. Diet studies of seabirds: a review of methods. Colonial Waterbirds, 9: 1–17.

Dierschke, A.-K., and Hüppop, O. 2003. Langfristige Veränderungen in der Ernährung von Silbermöwen (*Larus argentatus*) auf Helgoland unter dem Einfluss der Fischerei mit Vergleichen zur Heringsmöwe (*Larus fuscus*). Seevögel, 24: 3–15.

Erikstad, K. E., Fauchald, P., Tveraa, T., and Steen, H. 1998. On the cost of reproduction in long-lived birds; the influence of environmental variability. Ecology, 79: 1781–1788.

Finney, S. K., Wanless S., Harris, M. P., and Monaghan, P. 2001. The impact of gulls on puffin reproductive performance: an experimental test of two management strategies. Biological Conservation, 98: 159–165.

Freeman, A. N.D., and Smith, P. J. 1998. Iso-electric focusing and the identification of fisheries' waste in the diet of Westland petrels (*Procellaria westlandica*). New Zealand Journal of Marine and Freshwater Research, 32: 177–180

Frengen, O., and Thingstad, P. G. 2002. Mass occurrence of sandeels (*Ammodytes* spp.) causing aggregations of diving ducks. Fauna norvegica, 22: 32–36.

Furness, R. W. 1978. Kleptoparasitism by Great Skuas (*Catharacta skua* Brünn) and Arctic Skuas (*Stercorarius parasiticus* L.) at a Shetland seabird colony. Animal Behaviour, 26: 1167–1171.

Furness, R. W., and Todd, C. M. 1984. Diets and feeding of Fulmars *Fulmarus glacialis* during the breeding season: a comparison between St. Kilda and Shetland colonies. Ibis, 126: 379–387.

Gagliardi, A., Martinoli, A., Wauters, L., and Tosi, G. 2003. A floating platform: a solution to collecting pellets when cormorants roost over water. Waterbirds, 26: 54–55.

Garthe, S., Grémillet, D., and Furness, R. W. 1999. At-sea-activity and foraging efficiency in chick-rearing northern gannets (*Sula bassana*): a case study in Shetland. Marine Ecology Progress Series, 185: 93–99.

Garthe, S. Kubetzki, U., Hüppop, O., and Freyer, T. 1999. Zur Ernährungsökologie von Herings-, Silber- und Sturmmöwe (*Larus fuscus*, *L. argentatus* und *L. canus*) auf der Nordseeinsel Amrum während der Brutzeit. Seevögel, 20: 52–58.

Garthe, S., Montevecchi, W. A., Chapdelaine, G., Rail , J. F., and Hedd, A. 2007. Contrasting foraging tactics of seabirds breeding in different oceanographic domains. Marine Biology: in press.

González-Solís, J., Oro, D., and Pedrocchi, V. 1997. Bias associated with diet samples in Audouin's gulls. Condor, 99: 773–779.

Granadeiro, J. P, Monteiro, L. R., Silva, M. C., and Furness, R. W. 2002. Diet of common terns in the Azores, Northeast Atlantic. Waterbirds, 25: 149–155.

Grémillet, D., and Argentin, G., 1998. Cormorans et pêcheries autour des Iles Chausey [Cormorants, Shags and fisheries in the Chausey Islands area]. Le Cormoran, 10: 196–202.

Grémillet, D., and Plös, A. 1994. The use of stomach temperature records for the calculation of daily food intake in cormorants. Journal of Experimental Biology, 189: 105–115.

Grover, J. J., and Olla, B. L. 1983. The role of the rhinoceros auklet (*Cerorhinca monocerata*) in mixed-species feeding assemblages of seabirds in the Strait of Juan de Fuca, Washington. Auk, 100: 979–982.

Grunsky-Schöneberg, B. 1998. Brutbiologie und Nahrungsökologie der Trottellumme (*Uria aalge* Pont.) auf Helgoland. Ökologie der Vögel (Ecology of Birds), 20: 217 274.

Hald-Mortensen, P. 1995. Danske skarvers (*Phalacrocorax carbo sinensis*) fødevalg i 1992–1994 [The food of Danish Cormorants in 1992–1994 based on analyses of pellets from 23 different breeding colonies and two roosting sites]. Miljø og Energiministeriet Skov og Naturstyrelsen, 418p.

Halley, D. J., Harrison, N., Webb, A., and Thompson, D. R. 1995. Seasonal and geographical variations in the diet of common guillemots *Uria aalge* off western Scotland. Seabird, 17: 12–20.

Harding, A. M.A., van Pelt, T. I., Piatt, J. F., and Kitaysky, A. S. 2002. Reduction of provisioning effort in response to experimental manipulation of chick nutritional status in the horned puffin. Condor, 104: 842–847.

Härkönen, T. 1986. Guide to the otoliths of the bony fishes of the northeast Atlantic. Danbiu ApS., Hellerup, 256 pp.

Harris, M. P., and Wanless, S. 1993. The diet of shags *Phalacrocorax aristotelis* during the chick-rearing period assessed by three methods. Bird Study, 40: 135–139.

Harris, M. P., and Wanless, S. 1995. The food consumption of young common murres (*Uria aalge*) in the wild. Colonial Waterbirds, 18: 209–213.

Hedd, A., and Montevecchi, W. A. 2006. Diet and trophic position of Leach's Storm-Petrel during breeding and molt, inferred from stable isotope analysis of feathers. Marine Ecology Progress Series, 322: 291–301.

Hislop, J. R.G., Harris, M. P., and Smith, J. G.M. 1991. Variation in the calorific value and total energy content of the lesser sandeel (*Ammodytes marinus*) and other fish preyed on by seabirds. Journal of Zoology (London), 224: 501–517.

Hobson, K. A. 1993. Trophic relationships among high Arctic seabirds: insights from tissue-dependent stable-isotope models. Marine Ecology Progress Series, 95: 7–18.

Hobson, K. A., and Montevecchi, W. A. 1991. Stable isotopic determination of the trophic relationships of Great Auks. Oecologia, 87: 528–531.

Hobson, K. A., Piatt, J. F., and Pitochelli, J. 1994. Using stable isotopes to determine seabird trophic level relationships. Journal of Animal Ecology, 63: 786–798.

Hobson, K. A., and Welch, H. E. 1992. Determination of trophic relationships within a high Arctic marine food web using $\delta^{13}C$ and $\delta^{15}N$ analysis. Marine Ecology Progress Series, 84: 9–18.

Hüppop, O., Fründt, A. 2002. Zur Speiballen-Produktion freilebender Kormorane (*Phalacrocorax carbo*) im Winter. Jber. Institut Vogelforschung, 5: 11.

Iverson, S. J., Field, C., Bowen, W. D., and Blanchard, W. 2004. Quantitative fatty acid signature analysis: a new method of estimating predator diets. Ecological Monographs, 74: 211–235.

Johnson, J. H., Ross, R. M., and Smith, D. R. 1997. Evidence of secondary consumption of invertebrate prey by double-crested cormorants. Colonial Waterbirds, 20: 547–551.

Jost, O. 1975. Fisch-otolithen in Speiballen der Wasseramsel. Natur ünd Museum, 105: 283–286.

Käkelä, A., Crane, J., Votier, S. C., Furness, R. W., and Käkelä, R. 2006. Fatty acid signatures as indicators of diet in great skuas Stercorarius skua, Shetland. Marine Ecology Progress Series, 319: 297–310.

Käkelä, A, Furness R. W., Kelly, A., Strandberg, U., Waldron, S., and Käkelä, R. 2007. Fatty acid signatures and stable isotopes as dietary indicators in North Sea seabirds. Marine Ecology Progress Series ms. 6763.

Käkelä, R., Käkelä, A., Kahle, S., Becker, P. H., Kelly, A., and Furness, R. W. 2005. Fatty acid signatures in plasma of captive herring gulls as indicators of demersal or pelagic fish diet. Marine Ecology Progress Series, 293: 191–200.

Keijl, G., Roomen, M. Van, and Veldhuijzen van Zanten, H. 1986. Voedselecologie van de Stormmeeuw (*Larus canus*) te Schoorl 1986: Voedselkeuze en fourageerritme in de periode dat de jongen worden grootgebracht. Inst. lerarenopl., Hogeschool Holland, sektie biologie, Diemen 64 pp.

Kennedy, G. J.A., and Greer, J. E. 1988. Predation by cormorants, *Phalacrocorax carbo* L., on the salmonid populations of an Irish river. Aquaculture and Fisheries Management, 19: 159–170.

Kingston, S., O'Connell, M., and Fairley, J. S. 1999. Diet of otters *Lutra lutra* on Inishmore, Aran Islands, west coast of Ireland. Biological Environment Proceedings of the Royal Irish Academy, 99B: 173–182.

Kube, J. 1996. The ecology of macrozoobenthos and seaducks in the Pomeranian Bay. Meereswissenschaftliche Berichte, 18.

Kubetzki, U., Garthe, S., and Hüppop, O. 1999. The diet of common gulls *Larus canus* breeding on the German North Sea coast. Atlantic Seabirds, 1: 57–70.

Kubetzki, U., and Garthe, S. 2003. Distribution, diet and habitat selection by four sympatrically breeding gull species in the south-eastern North Sea. Marine Biology, 143: 199–207.

Larson, K., and Craig, D. 2006. Digiscoping vouchers for diet studies in bill-load holding birds. Waterbirds, 29: 198–202.

Lawson, J. W., Anderson, J. T., Dalley, E. L., and Stenson, G. B. 1998. Selective foraging by harp seals *Phoca groenlandica* in nearshore and offshore waters of Newfoundland, 1993 and 1994. Marine Ecology Progress Series, 163: 1–10.

Leopold, M. F., Baptist, H. J.M., Wolf P. A., and Offringa, H. 1995. De Zwarte Zeeëend *Melanitta nigra* in Nederland. Limosa, 68: 49–64.

Leopold, M. F., and Damme, C. J.G. van. 2003. Great cormorants *Phalacrocorax carbo* and polychaetes: can worms sometimes be a major prey of a piscivorous seabird? Marine Ornithology, 31: 75–79.

Leopold, M. F., Damme, C. J.G. van, and Veer, H. W. van der, 1998. Diet of cormorants and the impact of cormorant predation on juvenile flatfish in the Dutch Wadden Sea. Journal of Sea Research, 40: 93–107.

Leopold, M. F., Damme, C. J.G. van, Philippart, C. J.M., and Winter, C. J.N. 2001. Otoliths of North Sea fish: interactive guide of identification of fish from the SE North Sea, Wadden Sea and adjacent fresh waters by means of otoliths and other hard parts. CD-ROM, ETI, Amsterdam.

Leopold, M. F., van Elk, J. F., and van Heezik, Y. M., 1996. Central place foraging in Oystercatchers *Haematopus ostralegus*: can parents that transport mussels *Mytilus edulis* to their young profit from size selection?. Ardea, 84A: 311–325.

Leopold, M. F., Spannenburg, P. C., Verdaat, H. J.P., and Kats, R. K.H. (submitted). Identification and size estimation of *Spisula subtruncata* and *Ensis americanus* from shell fragments in stomachs and faeces of common eiders *Somateria mollissima* and Common Scoters *Melanitta nigra*. Atlantic Seabirds.

Lorentsen, S.-H., and Anker-Nilssen, T. 1999. Diet of common murres wintering in northern Skagerrak during 1988–1990: variation with sex, age and season. Waterbirds, 22: 80–89.

Ludynia, K., Garthe, S., and Luna-Jorquera, G. 2005. Seasonal and regional variation in the diet of the Kelp Gull in Northern Chile. Waterbirds, 28: 359–365.

Lønne, O. J., and Gabrielsen, G. W. 1992. Summer diet of seabirds feeding in sea-ice-covered waters near Svalbard. Polar Biology, 12: 685–692.

Martini, E., Reichenbacher, B. 1993. Ungewöhnlicher Eintrag von Fisch-Otolithen in rezente Sedimente bei Helgoland durch Silbermöwen (Larus argentatus). Zitteliana, 20: 379–387.

Mayr, G. 2004. A partial skeleton of a new fossil loon (Aves, Gaviiformes) from the early Oligocene of Germany with preserved stomach content. Journal of Ornithology, 145: 281–286.

McCleery, R. H., and Sibly, R. M. (1986): Feeding specialization and preference in Herring Gulls. Journal of Animal Ecology, 55: 245–259.

Mehlum, F. 2001. Crustaceans in the diet of Adult Common and Brünnich's guillemots *Uria aalge* and *U. lomvia* in the Barents Sea during the breeding period. Marine Ornithology, 29: 19–22.

Meijering, M. P.D. 1954. Zur Frage der Variationen in der Ernährung der Silbermöwe, *Larus argentatus* Pont. Ardea, 42: 163–175.

Montevecchi, W. A., Birt-Friesen V. L., and Cairns, D. K. 1992. Reproductive energetics and prey harvest by Leach's Storm-Petrels in the northwest Atlantic. Ecology, 73: 823–832.

Mühlenberg, M. 1989. Freilandökologie. Quelle and Meyer, Heidelberg.

Nehls, G., 1989. Occurrence and food consumption of the common eider, *Somateria mollissima*, in the Wadden Sea of Schleswig Holstein. Helgoländer Meeresuntersuchungen, 43: 385–393.

Nehls, G., and Ketzenberg, C. 2002. Do eiders, *Somateria mollissima*, exhaust their food resources? A study on natural mussel *Mytilus edulis* beds in the Wadden Sea. Danish Review of Game Biology, 16: 47–61.

Niebuhr, V. 1983. Feeding strategies and incubation behavior of wild herring gulls: an experiment using operant feeding boxes. Animal Behaviour, 31: 708–717.

Noordhuis, R., and Spaans, A. L. 1992. Interspecific competition for food between herring *Larus argentatus* and lesser black-backed gulls *L. fuscus* in the Dutch Wadden Sea area. Ardea, 80: 115–132.

Olmos, V., Aragoneses, J., Echevarrias, J. L., and Oltra, R., 2000. Composición de la dieta e impacto del Cormorán grande (*Phalacrocorax carbo sinensis*) durante la invernada en las Salinas de Santa Pola, Alicante, este de España. Ardeola, 47: 227–236.

Ormerod, S. J., and Tyler, S. J., 1991. Exploitation of prey by a river bird, the dipper *Cinclus cinclus* (L.), along acidic and circumneutral streams in upland Wales (UK). Freshwater Biology, 25: 105–116.

Orians, G. H., and Pearson, N. E. 1979. On the theory of central place foraging. In: Horn, D. J., Mitchell, R. D., and Stairs, G. R. (eds). Analysis of Ecological Systems, Ohio State University Press: 154–177.

Ouwehand, J., Leopold, M. F., and Camphuysen, C. J. 2004. A comparative study of the diet of guillemots *Uria aalge* and razorbills *Alca torda* killed during the Tricolor oil incident in the south-eastern North Sea in January 2003. Atlantic Seabirds (special issue) 6: 147–166.

Piatt, J. F. 1990. The aggregative response of Common Murres and Atlantic Puffins to schools of Capelin. *In:* Sealy, S. G. (ed.). Auks At Sea. Studies in Avian Biology, 14: 36–51.

Pierce, G. J., Boyle, P. R., and Diack, J. S.W. 1991. Identification of fish otoliths and bones in faeces and digestive tracks of seals. Journal of Zoology (London), 224: 320–328.

Pierce, G. J., Diack, J. S.W., and Boyle, P. R. 1990. Application of serological methods to identification of fish prey in diets of seals and dolphins. Journal of Experimental Marine Biology and Ecology, 137: 123–140.

Pierotti, R., and Annett, C. A. 1987. Reproductive consequences of dietary specialization and switching in an ecological generalist. In: Kamil A. C., Krebs J. R., and Pulliam H. R. (eds): Foraging Behavior: 417–442. Plenum Press, New York.

Pons, J.-M. 1994. Feeding strategies of male and female Herring Gulls during the breeding season under various feeding conditions. Ethology, Ecology & Evolution, 6: 1–12.

Pons, J.-M., and Migot, P. 1995. Life-history strategy of the herring gull: Changes in survival and fecundity in a population subjected to various feeding conditions. Journal of Animal Ecology, 64: 592–599

Prime J. H., and Hammond P. S. 1987. Quantitative assessment of gray seal diet from fecal analysis. In: Huntley, A. C., Costa, D. P., G. A.J. Worthy, and M. A. Castellini (Eds): Approaches to Marine Mammal Energetics. Society for Marine Mammalogy Special Publication No. 1: 165–181.

Rand, R. W. 1960. The biology of guano producing seabirds. The distribution, abundance and feeding habits of the Cape Penguin, *Spheniscus demersus*, off the South-Western Coast of the Cape Province. Commerce and Industry, March 1960: 5–17.

Ratcliffe, N., Richardson, D., Lidstone Scott, R., Bond, P. J., Westlake, C., and Stennett, S. 1997. Host selection, attack rates and success rates for Black-headed Gull kleptoparasitism of terns. Colonial Waterbirds, 20: 227–234.

Rau, G. H., Sweeney, R. E., and Kaplin, I. R.1982. Plankton $^{13}C:^{12}C$ ratio changes with latitude: Differences between northern and southern oceans. Deep Sea Research, 29 (8A): 1035–1039.

Reed, J. Z., Tollit, D. J., Thompson, P. M., and Amos, W. 1997. Molecular scatology: the use of molecular genetic analysis to assign species, sex and individual identity to seal faeces. Molecular Ecology, 6: 225–234.

Rodway, M. S., and Cooke, F., 2002. Use of fecal analysis to determine seasonal changes in the diet of wintering Harlequin Ducks at a herring spawning site. Journal of Field Ornithology, 73: 363–371.

Rodway, M. S., and Montevecchi, W. A. 1996. Sampling techniquess for assessing the diets of Atlantic puffin chicks. Marine Ecology Progress Series, 144: 41–55.

Rowe, S., Jones, I. L., Chardine, J. W., Elliot, R. D., and Veitch, B. G. 2000. Recent changes in the winter diet of murres (*Uria* spp.) in coastal Newfoundland waters. Candian Journal of Zoology, 78: 495–500.

Russell, A. F., Wanless, S., and Harris, M. P. 1995. Factors affecting the production of pellets by Shags *Phalacrocorax aristotelis*. Seabird, 17: 44–49.

Ryan, P. G., and Jackson, S. 1986. Stomach pumping: Is killing seabirds necessary? Auk 103: 427–428.

Sanger, G. A., and Hatch, S. A. 1987. Diets of nestling tufted puffins (*Fratercula cirrhata*) in the Gulf of Alaska and Eastern Aleutian Islands in 1986. Alaska Fish Wildlife Research Centre Report.

Scheiffarth, G., 2001. The diet of Bar-tailed Godwits *Limosa lapponica* in the Wadden Sea: combining visual observations and faeces analyses. Ardea, 89: 481–494.

Shealer, D. A. 1998. Differences in diet and chick provisioning between adult Roseate and Sandwich Terns in Puerto Rico. Condor, 100: 131–140.

Schneider, D. C., and Hunt. G. L. Jr. 1982. Carbon flux to seabirds in waters with different mixing regimes in the southeastern Bering Sea. Marine Biology, 67: 337–344.

Skov, H. Durinck, J., and Danielsen, F. 1989. Sammenfaldende forekomst mellem Søkonger *Alle alle* og Glaskutling *Aphya minuta* i Flekkefjord, Sydnorge, november1988 [Spatial coincidence of Little Auk *Alle alle* and the gobiid *Aphya minuta* in the Flekkefjord, south Norway, November 1988]. Pelagicus, 4: 22–24.

Sonntag, N., and Hüppop, O. 2005. Snacks from the depth: summer and winter diet of Common Guillemots *Uria aalge* around the Island of Helgoland. Atlantic Seabirds, 7: 1–14.

Spaans, A. L. 1971. On the feeding ecology of the Herring Gull *Larus argentatus* Pont. in the northern part of the Netherlands. Ardea, 59: 73–188.

Stenhouse, I. J., and Montevecchi, W. A. 1996. Winter distribution and "wrecks" of Dovekies in the Northwest Atlantic. Sula, 10: 219–228.

Suter, W., and Morel, P. 1996. Pellet analysis in the assessment of Great Cormorant *Phalacrocorax carbo* diet: reducing biases from otolith wear when reconstructing fish length. Colonial Waterbirds, 19: 280–284.

Swanson, G. A., Krapu, G. L., Bartonek, J. C., Serie, J. R., and Johnson, D. H. 1974. Advantages in mathematically weighting waterfowl food habits data. Journal of Wildlife Management, 38: 302–307.

Swennen, C. 1971. Het voedsel van de Groenpootruiter *Tringa nebularia* tijdens het verblijf in het Nederlandse Waddengebied. Limosa, 44: 71–83.

Swennen, C., 1976. Populatiestructuur en voedsel van de Eidereend *Somateria mollissima* in de Nederlandse Waddenzee. Ardea, 64: 311–371.

Sydeman, W. J., Hobson, K. A., Pyle, P., and McLaren, E. B. 1997. Trophic relationships among seabirds in central California: combined stable isotope and conventional dietary approach. Condor, 99, 327–336.

Taylor, A. J., and O'Halloran, J. 1997. The diet of the Dipper *Cinclus cinclus* as represented by faecal and regurgitate pellets: a comparison. Bird Study, 44: 338–347.

Tollit, D. J., and Thompson, P. M., 1996. Seasonal and between-year variations in the diet of harbour seals in the Moray Firth, Scotland. Canadian Journal of Zoology, 74: 1110–1121.

Tollit, D. J., Heaslip, S. G., Zeppelin, T. K., Joy, R., Call, K. A., and Trites, A. W. 2004. A method to improve size estimates of Walleye Pollock (*Theragra chalcogramma*) and Atka Mackerel (*Pleurogrammus monopterygius*) consumed by pinnipeds: digestion correction factors applied to bones and otoliths recovered from scats. Fisheries Bulletin, 102: 498–508.

Trauttmansdorff, J., and Wassermann G. 1995. Number of pellets produced by immature Cormorants *Phalacrocorax carbo sinensis*. Ardea, 83: 133–134.

Veen, J. 1977. Functional and causal aspects of nest distribution in colonies of the Sandwich Tern (*Sterna s. sandvicensis*). Behaviour Supplement, XX: 193 pp.

Veen, J., Peeters J., Leopold, M. F., van Damme, C. J.G., and Veen, T. 2003. Les oiseaux piscivores comme indicateurs de la qualité de l'environnement marin: suivi des effets de la pêche littorale en Afrique du Nord-Ouest. Alterra Report 666, 190 pp.

Walter, C. B., and O'Neill, E. 1986. Electrophoresis in the study of diets and digestive rates of seabirds. Comparative Biochemistry and Physiology, 84A: 763–765.

Walter, C. B., O'Neill, E., and Kirby, R. 1986: "ELISA" as an aid in the identification of fish and molluscan prey of birds in marine ecosystems. Journal of Experimental Marine Biology and Ecology, 96: 97–102.

Wanless, S., Daunt, F., Camphuysen, C. J., Greenstreet, S., and Humphreys, L. 2005. Setting the scene (4): seabirds in the Wee Bankie area. In: Camphuysen, C. J. (ed.) Understanding marine foodweb processes: an ecosystem approach to sustainable sandeel fisheries in the North Sea. IMPRESS Final Report. NIOZ-rapport 2005–5: 88–134.

Wanless, S., Wright, P. J., Harris, M. P., and Elston, D. A. 2004. Evidence for decrease in size of lesser sandeels *Ammodytes marinus* in a North Sea aggregation over a 30-yr period. Marine Ecology Progress Series, 279: 237–246.

Warham, J. 1996. The behavior, population biology and physiology of the petrels. Academic Press, San Diego.

Watt, J., Pierce, G. J., and Boyle, P. R., 1997. Guide to the identification of North Sea fish using premaxillae and vertebrae. ICES Cooperative Research Report No. 220, International Council for the Exploration of the Sea, Copenhagen.

Weimerskirch, H. 1998. How can a pelagic seabird provision its chick when relying on a distant food resource? Cyclic attendance at the colony, foraging decision and body condition in Sooty Shearwaters. Journal of Animal Ecology, 67: 99–109.

Wietfeld, J,. 1977. Untersuchungen am Speiballen der Silbermöwen (*Larus argentatus*) im Naturschutzgebiet Großer Knechtsand. Vogelwarte, 6: 221–229.

Wilson, R. P. 1984. An improved stomach pump for penguins and other seabirds. Journal of Field Ornithology, 55: 109–112.

Wilson, R. P., Cooper, J., and Plötz, J. 1992. Can we determine when marine enotherms feed: a case study with seabirds. Journal of Experimental Biology, 167: 267–275.

Wilson, R. P., Pütz, K., Grémillet, D., Culik, B. M., Kierspel, M., Regel, J., Bost, C. A., Lage, J., and Cooper, J. 1995. Reliability of stomach temperature changes in determining feeding characteristics of seabirds. Journal of Experimental Biology, 198: 1115–1135.

Wilson, R. P., Steinfurth, A., Ropert-Coudert, Y., Kato, A., and Kurita, M. 2002. Lip-reading in remote subjects: an attempt to quantify and separate ingestion, breathing and vocalistion in free-living animals using penguins as a model. Marine Biology, 140: 17–27.

Wilson, L. J., Daunt, F., and Wanless, S. 2004. Self-feeding and chick-provisioning diet differ in the Common Guillemot *Uria aalge*. Ardea, 92: 197–208.

Ydenberg, R. 1994. The behavioural ecology of provisioning in birds. Ecoscience, 1: 1–14.

Zijlstra, M., and van Eerden, M. R. 1995. Pellet production and the use of otoliths in determining the diet of cormorants *Phalacrocorax carbo sinensis*: Trials with captive birds. Ardea, 83: 123–131.

## 6 Ecological issues related to the circulation of pathogens and parasites in seabird populations

At the 2006 meeting, the group briefly discussed the need to review ecological issues related to the circulation of pathogens and parasites in seabird populations and this topic was put as a term of reference for the 2007 meeting. This chapter thus presents a first attempt on the topic, which includes a general outline of the subject and some details on specific issues. As a review of this topic is no simple, straight-forward task, WGSE needs to elaborate on this ToR on future meetings.

### 6.1 Introduction

Recent outbreaks of avian influenza (Olsen *et al.,* 2006) and West Nile viruses (Rappole and Hubalek 2003; Gerhardt, 2006) have highlighted the role that birds can play in the ecology of zoonotic diseases. The large population sizes of seabirds, their high mobility and their wide geographic distribution make them significant potential players in the ecology and epidemiology of diseases associated with wildlife, and in several instances they have been involved in major outbreaks (e.g. Olsen *et al.,* 2006; Herrmann *et al.,* 2006). The highly social breeding habits of seabirds and their high dispersive behaviour (Furness and Monaghan 1987) make them particular hosts for macro- and microparasites, with potential implications in terms of population ecology and epidemiology. The discrete distribution of seabirds among colonies and the large amount of existing knowledge about their breeding biology and at sea distribution also suggest that they could make good models to study the dynamics of interactions of vertebrate hosts with pathogens and parasites. Despite the recent active development of work on the ecology and evolution of host-parasite interactions (Grenfell and Dobson 1995; Hudson *et al.,* 2002; Frank, 2002; Thomas *et al.,* 2006), relatively little information is still available on seabird-parasite interactions and their epidemiological implications. We present here some elements on ecological issues related to the circulation of pathogens and parasites in seabird populations in order to identify gaps in our knowledge and potential avenues for research of basic and applied interest.

### 6.2 Seabirds as hosts for pathogens and parasites

Parasites can be defined in an ecological sense as organisms that live at the expense of others (Combes, 1995). This definition encompasses micro- and macro-parasites. Hosts and parasites are involved in intimate relationships that have implications at ecological and evolutionary scales (Combes, 1995). For instance, the costs associated with the parasitism of hosts by parasites can lead to differential survival or breeding success of individuals living in various ecological conditions, but the long-term interactions between hosts and parasites populations will also lead to the coevolution of their genomes (Combes, 1995). Hosts are mortal, so parasites face the need being transferred between hosts, which is a crucial issue as the probability of transmission is assumed to be linked to the virulence of the parasite strain. When competing strains invade a host, it is often assumed that the most virulent is the one able to transmit the fastest. This has direct implication when we consider the interactions between the life histories of hosts and parasites. An important issue related to the intimate nature of the relationships between hosts and parasites is the fact that parasites can be more or less specialized in terms of the host species they can exploit. Some parasites can exploit different hosts, which can have key epidemiological implications when some host species play the role of reservoirs for agents that can lead to diseases in others.

As warm-blooded vertebrates, seabirds are hosts of a large suite of pathogens and parasites (Hubalek 1994). A series of the life history and ecological characteristics of seabirds make

them particular hosts compared to other vertebrates, especially regarding the probability that parasites are transmitted among hosts breeding within a given geographic area (Table 6.1).

**Table 6.1. Seabirds as hosts for parasites: some implications of their life histories and ecology.**

| LIFE HISTORY PARTICULARITIES | IMPLICATIONS | EXAMPLES |
|---|---|---|
| Breeding site fidelity | Spatial structure of host-parasite interactions | Patterns of infestation ny nest dwelling parasites (ticks, fleas) (Rothschild and Clay 1957; Boulinier *et al.,* 1996) |
| Long-lived | Long temporal scale of some interactions (immunity, maternal effects…) | Interannual persistence and maternal transfer of antibodies against Lyme disease agent *Borrelia burgdorferi* (Gasparini *et al.,* 2001; Staszewski *et al.,* in press) |
| Colonial breeding | Possible high transmission among individuals | Avian influenza in Common terns *Sterna hirundo* (Olsen *et al.,* 2006) |
| Breeding with other species | Potentially for host transfer and/or host sharing (mixed colonies) | Evidence of host races for the seabird tick *Ixodes uriae* (McCoy *et al.,* 2005b) |
| Migratory and highly dispersive | Potential dispersal of parasites and their genes, although often constrained to the sea environment | Marine cycle of Lyme disease agent *Borrelia burgdorferi* (Olsen *et al.,* 1995) |
| Seasonal reproduction | Potential cyclicity in the interactions with parasites (Altizer *et al.,* 2006) | Cyclic exposure to nest dwelling parasites and associated microparasites |
| *Piscivorous / carnivorous /scavengers* | Potential role as hosts for complex endoparasite cycles and transmission of microparasites among hosts | Role in *Salmonella* epidemiology (Monaghan *et al.,* 1985) |

Detailed reviews of the micro-pathogenic organisms associated with free-living seabirds can be found in Hubalek (1994, 2004) and more specialized papers present reviews for seabird viruses (Chastel, 1988) or parasites of auks (Muzaffar and Jones, 2004). Here, we do not review this literature *in extenso*, but we highlight some information and references about pathogens and parasites of seabirds to illustrate potentially significant ecological issues. It should be noted that some specific host-parasite systems have been the subject of much work, while very little information is available on others even if they may be of interest, notably because they have been involved in dramatic outbreaks. An illustration of that is the large amount of work that has been done in the 1970s and 1980s on seabird arboviruses (Nuttal 1984; Chastel *et al.,* 1988), while more current work on seabird microparasites has been especially developing on Lyme disease bacteria *Borrelia burgdorferi sensu lato* since its identification in seabirds in 1993 (Olsen *et al.,* 1993), but little is known about the circulation of Avian influenza viruses in seabirds (Olsen *et al.,* 2006). Also, the fact that some detailed studies have been carried out at a time when genetic approaches were not much used increases the heterogeneity in the quality and quantity of information available on different host-parasite systems. In particular, population genetic approaches are increasingly used to infer the evolution of pathogens and parasites, but their use with natural host-parasite systems is still limited (De Meeûs *et al.,* in press).

### 6.2.1 Viruses

Many viruses are circulating in seabird populations (Hubalek, 2004), but we will focus in this current version of the chapter on two groups: arboviruses (often transmitted among seabirds by nest dwelling parasites such as ticks) and influenza A viruses. Regarding other types of viruses, one of particular interest because of its known pathogenic effects on birds is Newcastle Disease Virus, which is known to circulate in cormorants and shags, but usually in inland areas. It should also be noted that recent investigations on coronaviruses in wild birds clearly stress that little is known about the presence of several viruses in wild populations (Monceyron-Jonassen *et al.,* 2005).

Many Arboviruses have been identified in seabird species parasitized by soft and hard ticks (Clifford 1979; Nuttal 1984; Chastel 1988; Chastel *et al.,* 1990). These arboviruses are circulating at different latitudes, notably due to their associations with different ticks, such as *Ixodes uriae* in the North Atlantic (Main *et al.,* 1976; Spence *et al.,* 1985; Nunn *et al.,* 2006a). Where intensive work has been carried out, as for instance in Brittany, Western France, several of these arboviruses have been identified in the ectoparasite species parasitizing the same host species in the same area (e.g. *Ixodes uriae, Ornithodoros maritimus*), suggesting the natural exposure of some seabirds, such as kittiwakes, to a high diversity of viruses (Chastel *et al.,* 1981, 1987, 1990; Quillien *et al.,* 1986). In general, not much is known about the pathogenicity of these viruses, although evidence of massive failure of tern colonies have been associated with the detection of some viruses and other viruses have been reported to have pathogenic effects on humans (Converse *et al.,* 1975; Feare *et al.,* 1976; Clifford *et al.,* 1980).

Influenza viruses are circulating in seabird populations although knowledge is still limited. A recent study showed for instance that three out of 26 Common guillemots *Uria aalge* banded in the Baltic Sea tested positive for influenza A virus using RT-PCR (Wallesten *et al.,* 2005). Phylogenetic analyses further showed that five gene segments belonged to the American avian lineage of influenza A viruses, whereas three gene segments belonged to the Eurasian lineage. This indicates that avian influenza viruses may have a taxonomically wider reservoir spectrum than previously known and that naturally occurring chimeric avian influenza A viruses can include genes of American and Eurasian origin in Europe. Influenza A viruses have been known for a long time to circulate in duck populations and the close contact occurring between waterbirds species in some area calls for attention to the potential circulation of such viruses in seabirds (Muzaffar *et al.,* 2006).

### 6.2.2 Bacteria

Many bacteria are circulating in seabird populations (Hubalek, 2004). Among them, attention has especially been driven in recent years to Lyme disease agent *Borrelia burgdorferi sensu lato*, which was shown to circulate in seabirds only relatively recently (Olsen, 1993). Lyme disease is the main human arthropod borne disease in the northern hemisphere. It is due to various genospecies within the *Borrelia burgdorferi* sensu lato which are transmitted by tick bites to humans, usually by 'terrestrial' ticks such as *Ixodes ricinus* and *Ixodes scapularis*. Pathogenic genospecies of *Borrelia burgdorferi* sensu lato have been identified in seabirds (Olsen *et al.,* 1993, 1995; Gylfe *et al.,* 1998; Smith *et al.,* 2006; McCoy *et al.,* unpublished) and some of its epidemiological (Olsen, *et al.,* 1995; Gylfe *et al.,* 1998) and ecological implications have been investigated (Gasparini *et al.,* 2001, 2002; Staszewski *et al.,* in press).

### 6.2.3 Other parasites

Several other parasites are of importance for seabirds and they will be briefly reviewed in the next version of this chapter. This is notably the case of Fungi, Protozoa as well as macroparasites such as Ectoparasites (ticks, fleas, feather lice; Rothschild and Clay, 1957; Murray and Vestjens, 1967; Duffy, 1983; Guiguen, 1988) and Endoparasites (Trematodes and

cestodes notably, Threlfall, 1968). Some details about the seabird tick *Ixodes uriae* are provided in the rest of the chapter in relation to its important vector role for viruses and bacteria.

## 6.3 Factors affecting the circulation of parasites in seabird populations

### 6.3.1 Seabird population biology and parasites

As summarized in Table 6.1, a series of common characteristics of seabird species are likely to be important for their ecological and evolutionary interactions with parasites. The re-uses of nesting sites year after year and dense aggregations occurring at nesting colonies are obvious characteristics that are favourable for the maintenance of high parasite populations. Large numbers of ectoparasites can in particular be found on most seabird colonies (Guiguen, 1988).

Negative effects on reproductive success when parasite loads are high can be significant, with potential consequences for the local dispersal of breeders via mechanisms of differential dispersal and recruitment as a function of local breeding success (Boulinier and Danchin, 1996; Boulinier *et al.,* 2001; Gauthier-Clerc *et al.,* 2003). Currently, there is nevertheless no evidence of negative effects on adult survival, but very few attempts have been made to investigate this issue.

Population genetic investigations using neutral markers (microsatellites) have shown that gene flows between populations of the seabird tick *Ixodes uriae* are much more restricted than gene flows between its host populations at the same geographic scales (McCoy *et al.,* 2003, 2005a). An interesting result is also that the ecology and behaviour of the seabird host species may lead to different dispersal rates among colonies of the ticks; for instance, tick populations are much more structured between Black-legged kittiwake populations than between Atlantic puffins *Fratercula arctica* populations when investigated at the same geographical scale (McCoy *et al.,* 2003), which could be due to a higher tendency for prospecting individuals to disperse ticks among colonies in puffins than kittiwakes. Another important finding was that several seabird populations breeding in mixed colonies (i.e. in sympatry) do not appear to share the same *Ixodes uriae* tick populations as revealed by population genetic analyses of the ticks collected on different host (McCoy *et al.,* 2001, 2005b). This suggests a specialization of the tick for its hosts, which should have implications for the circulation of microparasites such as *Borrelia* and arboviruses in seabird populations.

### 6.3.2 Global change and the circulation of pathogens and parasites in seabird populations

Human activities at various spatial scales can have dramatic consequences for the circulation of parasites in seabird populations and between free-ranging birds and humans and domesticated populations. Refuse dumps and fish factories have been identified as foci of bacteria transmission (e.g. *Salmonella*) involving gulls (Monaghan *et al.,* 1985; Nesse *et al.,* 2005). Concerns of exchange of parasites between intensive animal production facilities (notably of poultry) and free-ranging birds have been raised following the recent outbreaks of Avian influenza (Olsen *et al.,* 2006). Ecotourism to remote areas hosting large seabird colonies has the potential to lead to exchange of parasites (Wallensten *et al.,* 2006).

The effects of climate change on the circulation of parasites, notably via induced changes in the distribution of vectors, could be important. Much work is done on the potential effects of climate change on the distribution of tick species responsible for the terrestrial cycle of Lyme disease, such as *Ixodes scapularis* (Ogden *et al.,* 2005), and knowledge of the temperature and humidity tolerance of the seabird tick *Ixodes uriae* could be important in this context. It should nevertheless be noted that the species shows clear adaptation to dramatic variations of its

living environmental conditions (Murray and Vestjens, 1967; Lee and Baust, 1982; Barton *et al.,* 1996; Benoit *et al.,* 2007).

## 6.4 Implications of the circulation of pathogens and parasites in seabird populations

### 6.4.1 Pathogens and parasites and threats to seabird populations and biodiversity

The role of parasitism in the ecology of natural bird populations has attracted much interest in the last two decades, notably in the field of behavioural and population biology (Lloye and Zuk, 1991; Clayton and Moore, 2007), but those factors have not often been clearly identified as major threats to seabird populations. This is not to say that parasites do not play a significant role in the ecology of seabirds, as for instance breeding failure and dispersal among sub-colonies have been attributed to high local levels of ectoparasites in various species (Boulinier *et al.,* 2001).

As pathogens and parasites have the potential to spread rapidly among dense populations, such as seabird colonies, and as they have the potential to lower dramatically a key demographic parameter such as adult survival (to which seabird populations are highly sensitive), they nevertheless represent a factor that has the potential to be important.

An interesting issue relates to the question of whether the circulation of parasites in seabird populations is more or less independent of the circulation of parasites in other animals and humans. *Borrelia burgdoferi* may have been circulating in seabird populations for thousands of years independently of the terrestrial cycle which involve most human cases (Olsen *et al.,* 1995; Gauthier-Clerc *et al.,* 1999), but alternatively the development of ecotourism in remote areas with large seabird colonies raised the question of the risk of exposure of seabirds to new parasites, such as in the case of penguins in Antarctica (Wallensten *et al.,* 2006). It is nevertheless often hard to know whether parasites were native or introduced (Gauthier-Clerc *et al.,* 2002). As we have seen above, global change could lead to increased exposure of individuals and populations to parasites with which they have not evolved, which could have major ecological and epidemiological implications.

### 6.4.2 Seabird parasites and human diseases

Seabirds are carriers of a few diseases that can transmit to humans (zoonoses), but as humans are rarely in close contact with them this poses relatively little health concern. On islands where humans are still exploiting seabirds for food, cases of exposure of humans to zoonotic agents have been reported, such as seropositivity of bird hunters against *Borrelia burgdoferi* in the Faroe Islands (Gylfe *et al.,* 1998). Seemingly, biologists handling seabirds and working in areas infested by seabird ectoparasites may be at risk of exposure. The tick *Ixodes uriae* can for instance bite humans; although the fast detection and removal of feeding ticks is likely to often preclude the transmission of a bacteria such as Lyme disease to humans (transmission via a tick bite only occurs after several days of attachment to the host). One other instance that can lead to risks of exposure of humans to parasites carried by seabirds is in the context of rehabilitation programmes associated with the care for e.g. oiled birds (Steele *et al.,* 2005).

Due to their high dispersive behaviour and colonial habits, seabirds have the potential to disperse microparasites at large scales. Their role in the epidemiology of zoonoses and notably the emergence of infectious diseases, is nevertheless much constrained by the low contact between seabirds and wild terrestrial vertebrates (e.g. mammals, waterfowls, passerine birds), domesticated animals (livestock) or humans. As we have seen above, global changes may nevertheless increase the contact between host-parasite systems, which could lead to the rapid

changes in their dynamics. A better knowledge of the dynamics of these complex systems is thus required.

## 6.5 Perspectives

### 6.5.1 Monitoring of pathogens and parasites

As part of monitoring programmes of seabird populations involving the handling of seabirds for marking and/or diet sampling, samples could easily be taken to enable the tracking of the circulation of parasitic agents in populations at small and wide spatial scales. Such investigations can rely on molecular techniques (PCR methods) or the detection of antibodies in plasma or sera samples using immunological assays (ELISA, Western blots). It should be noted that as seabirds are long-lived, serological approaches, which rely on the detection of antibodies which may last for months and years in the host individuals after their exposure to an antigen, may provide limited information about the timing of exposure of individuals to antigens (Staszewski *et al.,* in press). If adults are difficult to capture, maternal antibodies can be detected in the egg yolk and the plasma of young chicks as these are available in an amount proportional to that circulating in the female plasma (Gasparini *et al.,* 2002). It should nevertheless be noted that the presence of antibodies in a five day old chick does not mean that the parasite is currently circulating in the host population as the mother may have been exposed to the parasites just once several years before, which may have been sufficient for a strong humoral immune response to be initiated (e.g. antibodies against *Borrelia burgdorferi* sl in Black-legged kittiwakes *Rissa tridactyla*; Staszewski *et al.,* in press).

The culture and sequencing of DNA from micro-organisms can enable researchers to investigate phylodynamic issues (origin of the strain and relationship with other strains) provided there are enough samples available on the studied organism. Such approaches should provide crucial elements for a better understanding of the processes responsible for the circulation of parasites in natural seabird populations. In the case of vector-born parasite, such approaches can also be used on the vector to disentangle factors responsible for strain differentiation (De Meeûs *et al.,* in press).

As for any monitoring programme of biodiversity, the why, what and how should be considered seriously before engaging in the gathering of information and samples (Yoccoz *et al.,* 2001). Particular attention to the spatial variability expected at different spatial (e.g. within and among colonies) and temporal scales (seasonality and inter-year variability) should be taken. In particular, the small numbers of samples often screened in past published studies sometime limit the inference that can be made about factors potentially affecting parasite dynamics. Relatively descriptive studies investigating the potential evidence of various parasites are nevertheless useful to plan more extensive surveys (Gauthier-Clerc *et al.,* 2002, Uhart *et al.,* 2003).

### 6.5.2 Perspective of research

As seen above, a lot of information is now available via classical parasitology studies as well as more recent work carried out in more ecological and evolutionary framework. The use of population and evolutionary genetics should enable us to get a much better understanding of the circulation of parasitic agents. The current development of work at the interplay between immunology and ecology (immuno-ecology; Sheldon and Verhulst, 1996; Viney *et al.,* 2005) could also provide interesting progress on some questions. For instance, the need to consider age-related effects and host immunity when undertaking quantitative studies of tick-borne pathogen transmission is becoming obvious (Gasparini *et al.,* 2001; Nunn *et al.,* 2006b). Interactions between host immunity and life history issues such as stress involved with migration should also be considered (Gylfe *et al.,* 2000).

The development of these perspectives of research should thus rely on data from monitoring programmes and surveillance activities conducted at large scales, but also on the development of specific case studies, which are necessary to complete our understanding of the ecology and epidemiology of seabird-parasite systems. Due to their relatively simple and spatially discrete structure, some seabird-parasite systems provide excellent opportunity for addressing questions of broad interest on the ecology and evolution of host-parasite interactions, as well on the effects of global changes on the risk of emerging infectious diseases.

## 6.6  References

Altizer, S., Dobson, A. P., Hosseini, P., Hudson, P. J., Pascual, M., and Rohani, P. 2006. Seasonality and the dynamics of infectious diseases. Ecology Letters, 9: 467–484.

Barton, T. R., Harris, M. P., Wanless, S., and Elston, D. A. 1996. The activity periods and life cycle of the tick *Ixodes uriae* (Acari: *Ixodidae*) in relation to host breeding strategies. Parasitology, 112: 571–580.

Benoit, B., Yoder, J. A., Lopez-Martinez, G., Elnitsky, M. A., Lee, R. E. Jr., and Denlinger, D. L. 2007. Habitat requirements of the seabird tick, *Ixodes uriae* (Acari: Ixodidae), from the Antarctic Peninsula in relation to water balance characteristics of eggs, nonfed and engorged stages. Journal of Comparative Physiology B: Biochemical, Systemic, and Environmental Physiology, 177: 205–215.

Boulinier, T., Ives, A. R., and Danchin, E. 1996. Measuring aggregation of parasites at different host population levels. Parasitology, 112: 581–587.

Boulinier, T., and Danchin, E. 1996. Population trends in Kittiwake *Rissa tridactyla* colonies in relation to tick infestation. Ibis, 138: 326–334.

Boulinier, T., Sorci, G., Monnat, J.-Y., and Danchin, E. 1997. Parent-offspring regression suggests heritable susceptibility to ectoparasites in a natural population of kittiwake *Rissa tridactyla*. Journal of Evolutionary Biology, 10: 77–85.

Boulinier, T., McCoy, K. D., and Sorci, G. 2001. Dispersal and parasitism. Pages 169–179 *in* J. Clobert, E. Danchin, A. Dhondt, and Nichols, J. D., editors. Dispersal. Oxford University Press, Oxford, UK.

Chastel, C. 1988. Tick-borne virus infections of Marine Birds. In Advances in disease vector research, Harris, K. F. (ed.): 25–60. Springer-Verlag: New-York.

Chastel, C., Monnat, J. Y., Le Lay, G., Guiguen, C., Quillien, M. C., and Beaucournu, J. C. 1981. Studies on Bunyaviridae including Zaliv Terpeniya virus isolated from *Ixodes uriae* ticks (Acarina, Ixodidae) in Brittany, France. Arch. Virol., 70: 357–366.

Chastel, C., Monnat, J. Y., Le Lay, G., and Balouet, G. 1987. Infestation et hyperinfestation de la Mouette tridactyle *Rissa tridactyla* L. par des tiques (*Ixodes (Ceratixodes) uriae*, *Ornithodoros (Alectorobius) maritimus*): conséquences pathologiques. Annals de Parasitologie Humaine et Comparée 1967, 62: 492–504.

Chastel, C., Le Lay, G., and Monnat, J.-Y. 1990. Isolation of Soldado virus from a seabird (*Rissa tridactyla* L.) in France. Acta virologica, 34: 109.

Clayton, D. H., and Moore, J. 1997. Host-parasite evolution. General principles and avian models. Oxford University Press: Oxford.

Clifford, C. M. 1979. Tick-born viruses of seabirds. In Arctic and tropical arboviruses. Kurstak, E. (ed): 83–100. Academic Press: New York.

Clifford, C. M., Hoogstraal, H., Radovsky, F. J., Stiller, D., and Keirans, J. E. 1980. *Ornithodoros (Alectorobius) amblus* (Acarina: Ixodoidea: Argasidae): identity, marine bird and human hosts, virus infections, and distribution in Peru. Journal of Parasitology, 66: 312–323

Combes, C. 1995. *Ecologie des interactions durables*. Masson, Paris, France.

Comstedt, P., Bergström, S., Olsen, ´B., Garpmo, U., Marjavaara, L., Mejlon, H., Barbour, A. G., and Bunikis, J. 2006. Migratory passerine birds as reservoirs of Lyme borreliosis in Europe. Emerging Infectious Diseases, 12: 1087–1095.

Converse, J. D., Hoogstraal, H., Moussa, M. I. *et al.* 1975. Soldado virus (Hughes group) associated from *Ornithodoros* (*Alectorobius*) *capensis* (Ixodoidea: Argasidae) infesting Sooty tern colonies in the Seychelles, Indian Ocean. American Journal of Tropical Medicine and Hygene, 24: 1010–1018.

Danchin, E. 1992. The incidence of the tick parasite *Ixodes uriae* in Kittiwake Rissa tridactyla colonies in relation to the age of the colony, and a mechanism of infecting new colonies. Ibis, 134: 134–141.

De Meeûs, T, McCoy, K. D., Prugnolle, F., Durand, P., Chevillon, C., Hurtrez-Boussès, S., and Renaud, F. 2006. Population genetics and molecular epidemiology or how to "débusquer la bête". Infection, Genetics & Evolution, in press.

Duffy, D. C. 1983. The ecology of tick parasitism on densely nesting peruvian seabirds. Ecology, 64: 110–119.

Feare, C. J. 1976. Desertion and abnormal development in a colony of Sooty Terns Sterna fuscata infested by virus-infected ticks. Ibis, 118: 112–115.

Frank, S. A. 2002. Immunology and evolution of infectious disease. Princeton University Press, Princeton, New Jersey, USA.

Furness, R. W., and Monaghan, P. 1987. Seabird ecology. Backie, Glasgow.

Gerhardt, R. 2006. West Nile virus in the United States (1999–2005), Journal of the American Animal Hospital Association, 42: 170–177.

Gasparini, J., McCoy, K. D., Haussy, C., Tveraa, T., and Boulinier, T. 2001. Induced maternal response to the Lyme disease spirochaete *Borrelia burgdorferi sensu lato* in a colonial seabird, the kittiwake *Rissa tridactyla*. Proceedings of the Royal Society B, 268: 647–650.

Gasparini, J., McCoy, K. D., Tveraa, T., and Boulinier, T. 2002. Related concentrations of specific immunoglobulins against the Lyme disease agent *Borrelia burgdorferi* s.l. in eggs, young and adults of the kittiwake (*Rissa tridactyla*). Ecology Letters, 5: 519–524.

Gauthier-Clerc, M., Jaulhac, B., Frenot, Y., Bachelard, C., Monteil, H., Le Maho, Y., and Handrich, Y. 1999. Prevalence of *Borrelia burgdorferi* (the Lyme disease agent) antibodies in king penguin *Aptenodytes patagonicus* in Crozet Archipelago. Polar Biology, 22: 141–143.

Gauthier-Clerc, M., Eterradossi, N., Toquin, D., Guittet, M., Kuntz, G., and Le Maho, Y. 2002. Serological survey of the king penguin, *Aptenodytes patagonicus* in Crozet Archipelago for antibodies to infectious bursal disease, infleunza A and Newcastle disease viruses. Polar Biology, 25: 316–319.

Gauthier-Clerc, M., Moangin, S., Le Bohec, C., Gendner, J.-P., and Le Maho, Y. 2003. Comparision of behaviour, body mass, haematocrit level, site fidelity and ssurvival between infested and non-infested king penguin Aptenodytes patagonicus by ticks Ixodes uriae. Polar Biology, 26: 379–382.

Grenfell, B. T., and Dobson, A. P. 1995. *Ecology of infectious diseases in natural populations*. Cambridge University Press, Cambridge, UK.

Guiguen, C. 1988. Anthropozoonoses et oiseaux marins: Contribution à l'étude des ectoparasites hématophages des espèces nicheuses sur les côtes françaises continentales et insulaires. Thèse pour l'obtention du diplôme d'Etat de Docteur en Biologie Humaine. Faculté de Médecine de Marseille.

Gylfe, A., Bergström, S., Lundstrom, J., and Olsen, B. 2000. Epidemiology: reactivation of *Borrelia* infection in birds. Nature, 403: 724–725.

Gylfe, Å., Olsen, B., Staševičius, D., Ras, N. M., Weihe, P., Noppa, L., Östberg, Y., Baranton, G., and Bergström, S. 1999. Isolation of Lyme disease *Borrelia* from puffins (*Fratercula arctica*) and seabird ticks (*Ixodes uriae*) on the Faeroe Islands. Journal of Clinical Microbiology, 37: 890–896.

Herrmann, B., Persson, H., Jens-Kjeld Jensen, J.-K., Debes Joensen, H., Klint, M., and Olsen, B. 2006. *Chlamydophila psittaci* in Fulmars, the Faroe Islands. 2006. Emerging Infectious Diseases, 12: 330–332.

Holt, R., and Boulinier, T. 2005. Ecosystem and parasitism: the spatial dimension. In *Ecosystem and parasitism*. Thomas, F., Renaud, F., and Guegan, J.-F. (eds). Oxford University Press, Oxford : 68–84.

Hubalek, Z. 1994. Pathogenic microorganisms associated with free-living birds (A review). Acta Scientiarum Naturalium Academiae Scientiarum Bohemicae, Bron 5: 1–74.

Hubalek Z. 2004. An annotated checklist of pathogenic microorganisms associated with migratory birds . Journal of Wildlife Diseases, 40: 639–659.

Hudson, P. J., Rizzoli, A., Grenfell, B. T., Heesterbeek, H., and Dobson, A. P. 2002. The ecology of wildlife diseases. Oxford University Press, Oxford, UK.

King, K. A., Keith, J. O., and Keirans, J. E. 1977a. Ticks as a factor in nest desertion of California Brown Pelicans. Condor, 79: 507–509.

King, K. A., Blankinship, D. R., Paul, R. T., and Rice, R. C.A. 1977 b. Ticks as a factor in the 1975 nesting failure of Texas Brown Pelican. Wilson Bulletin, 89: 157–158.

Komar, N. 2001. West Nile virus surveillance using sentinels birds. Annals of the New York Academy of Sciences, 951: 58–73.

Lee, R. E.Jr., and Baust, J. G. 1982. Respiratory metabolism of the antarctic tick, *Ixodes uriae*. Comparative Biochemistry and Physiology, 72A: 167–171.

Loye, J. E., and Zuk, M. 1991. Bird-Parasite Interactions. Ecology, Evolution and Behaviour. Oxford University Press: New York.

Main, A. J., Shope, R. E., and Wallis, R. C. 1976. Cape Wrath: a mew Kemerovo group orbivirus from *Ixodes uriae* (Acari: Ixodidae) in Scotland. Journal of Medical Entomology, 13: 304–308.

Mehl, R., and Traavik, T. 1983. The tick *Ixodes uriae* (Acari, Ixodites) in seabird colonies in Norway. Fauna norv. Ser. B., 30: 94–107.

McCoy, K. D., Boulinier, T., Chardine, J. W., Danchin, E., and Michalakis, Y. 1999. Dispersal and distribution of the tick *Ixodes uriae* within and among seabird host populations: the need for a population genetic approach. Journal of Parasitology, 85: 196–202.

McCoy, K. D., Boulinier, T., Schjoerring, S., and Michalakis, Y. 2002. Local adaptation of an ectoparasite *Ixodes uriae* to its seabird host. Evolutionary Ecology Research, 4: 441–456.

McCoy, K. D., Boulinier, T., Tirard, C., and Michalakis, Y. 2001. Host specificity of a generalist parasite: genetic evidence of sympatric host races in the seabird tick *Ixodes uriae*. Journal of Evolutionary Biology, 14: 395–405.

McCoy, K. D., Boulinier, T., Tirard, C., and Michalakis, Y. 2003. Host-dependent genetic structure of parasite populations: differential dispersal of seabird tick host races. Evolution, 57: 288–296.

McCoy, K. D., Tirard, C., and Michalakis, Y. 2003. Spatial genetic structure of the ectoparasite *Ixodes uriae* within breeding cliffs of its colonial seabird host. Heredity, 91: 422–429.

McCoy, K. D., Boulinier, T., and Tirard, C. 2005a. Comparative host-parasite population structures: disentangling prospecting and dispersal in the Black-legged kittiwake *Rissa tridactyla*. Molecular Ecology, 14: 2825–2838.

McCoy, K. D., Chapuis, E., Tirard, C., Boulinier, T., Michalakis, Y., LeBohec, C., LeMaho, Y., and Gauthier-Clerc, M. 2005b. Recurrent evolution of host-specialized races in a globally-distributed ectoparasite. Proceedings of the Royal Society of London B, 272: 2389–2395.

Monaghan, P., Shedden, C. B., Ensor. K. *et al.* 1985. *Salmonella* carriage by gulls (*Larus argentatus*) in the Clyde area of Scotland (U. K.) in relation to their feeding ecology. Journal of Applied Ecology, 22: 669–680.

Monceyron-Jonassen, C., Kofstad, T., Larsen, I.-L., Løvland, A., Handeland, K., Follestad, A., and Lillehaug, A. 2005. Molecular identification and characterization of novel coronavirus infecting graylag geese (*Anser anser*), feral pigeons (*Columbia livia*) and mallards (*Anas platyrhynchos*). Journal of General Virology, 86: 1597–1607.

Murray, M. D., and Vestjens, W. J.M. 1967. Studies on the ectoparasites of seals and penguins. III. The distribution of the tick *Ixodes uriae* White and the flea *Parapsyllus magellanicus* heardi De Meillon on Macquarie Island. Australian Journal of Zoology, 15: 715–725.

Muzaffar, S., and Jones, I. L. 2004. Parasites and diseases of the Auks (Alcidae) of the world and their ecology–a review. Marine Ornithology, 32: 121–146.

Muzaffar, S., Ydenberg, R. C., and Jones, I. L. 2006. Avian Influenza: An Ecological and Evolutionary Perspective for Waterbird Scientists. Waterbirds, 29: 243–257.

Nesse, L. L., Refsum, T., Heir, E., Nordby, K., Vardund, T., and Holstad, G. 2005. Molecular epidemiology of *Salmonella* spp. isolates from gulls, fish-meal factories, feed factories, animals and humans in Norway based on pulsed-field gel electrophoresis. Epidemiology and Infection, 133: 53–58.

Nunn, M. A., Barton, T. R., Wanless, S., Hails, R. S., Harris, M. P., and Nuttall, P. A. 2006. Tick-borne great island virus: identification of seabird host and evidence for co-feeding and viraemic transmission. Parasitology, 132: 233–240.

Nunn, M. A., Barton, T. R., Wanless, S., Hails, R. S., Harris, M. P., and Nuttall, P. A. 2006. Tick-borne Great Island Virus: (II) impact of age-related acquired immunity on transmission in a natural seabird host. Parasitology, 132: 241–253.

Nuttal, P. A. 1984. Tick-borne viruses in seabird colonies. Seabird 7: 31–41.

Ogden, N. H., Bigras-Poulin, M., O'Callaghan, C. J., Barker, I. K., Lindsay, L. R., Maarouf, A., Smoyer-Tomic, K. E., Waltner-Toews, D., and Charron, D. 2005. A dynamic population model to investigate effects of climate on geographic range and seasonality of the tick *Ixodes scapularis*. International Journal of Parasitiology, 35: 375–389.

Olsen, B., Jaenson, T. G.T., Noppa, L., Bukinis, J., and Bergström, S. 1993. A lyme borreliosis cycle in seabirds and *Ixodes uriae* ticks. Nature, 362: 340–342.

Olsen, B., Duffy, D. C, Jaenson, T. G.T., Gylfe, A., Bonnedahl, J., and Bergström, S. 1995. Transhemispheric echange of Lyme disease spirochetes by seabirds. Journal of Clinical Microbiology, 33: 3270–3274.

Olsen, B., Munster, V. J., Wallensten, A., Waldenström, J., Osterhaus, A. D.M. E., and Fouchier, R. A.M. 2006. Global patterns of Influenza A virus in wild birds. Science, 312: 384–388.

Quillien, M. C., Monnat, J. Y., Lay, G., Le Goff, F., Hardy, E., and Chastel, C. 1986. Avalon virus, Sakhalin group (Nairovirus, Bunyviridae) from seabird tick *Ixodes* (*Ceratixodes*) *uriae* White 1852 in France. Acta virologica. 30: 418–427.

Rappole, J. H., and Hubalek, Z. 2003. Migratory birds and West Nile virus, Journal of Applied Microbiology, 94: 47S-58S.

Rothschild, M., and Clay, T. 1957. Fleas, Flukes and Cuckoos. London: Colins.

Sheldon, B. C., and Verhulst, S. 1996. Ecological immunology: costly parasite defences and trade-offs in evolutionary ecology. Trends in Ecology and Evolution, 11: 317–321.

Smith, R. P., Bin Muzaffar, S., Lavers, J., Lacombe, E. H., Cahill, B. K., Lubelczyk, C. B., Kinsler, A., Mathers, A. J., and Rand, P. W. 2006. *Borrelia garinii* in seabird ticks (*Ixodes uriae*), Atlantic coast, North America. Emerging Infectious Diseases, 12: 1909–1912.

Spence, R. P., Harrap, K. A., and Nuttal, P. A. 1985. The isolation of Kemerovo group orbiviruses and uukuniemi group viruses of the family Bunyaviridae from Ixodes uriae ticks from the isle of May, Scotland. Acta Virologica, 29: 129–136.

Staszewski, V., McCoy, K. D., Tveraa, T., and Boulinier, T. in press. Interannual dynamics of antibody levels in naturally infected individuals of a long-lived colonial bird. Ecology.

Steele, M., Davies, C. R., Jones, L. D., Nuttall, P. A., and Rideout, K. 1990. Life history of the seabird tick, *Ixodes* (*Ceratixodes*) *uriae*, at St Abb's Head, Scotland. Acarologia, 31: 125–130.

Steele, C. M., Brown, R. N., and Botzler, R. G. 2005. Prevalences of zoonotic bacteria among seabirds in rehabilitation centers along the Pacific Coast of California and Washington, USA. Journal of Wildlife Diseases, 41: 735–744.

Thomas, F., Renaud, F., and Guegan, J.-F. eds. 2005. *Ecosystem and parasitism*. Oxford University Press, Oxford.

Threlfall, W. 1968. The helminth parasites of three species of gulls in Newfoundland. Canadian Journal of Zoology, 46: 827–830.

Uhart, M. M., Quintana, F., Karesh, W. B., and Braselton, W. E. 2003. Hematology, plasma biochemistry, and serosurvey for selected infectious agents in southern giant petrels from Patagonia, Argentina. Journal of Wildlife Diseases, 39: 359–365.

Viney, M. E., Riley, E. M., Buchanan, K. L. 2005. Optimal immune responses: immunocompetence revisited. Trends in Ecology and Evolution, 20: 665–669.

Wallensten, A., Munster, V. J., Elmberg, J., Osterhaus, A. D.M. E., Fouchier, R. A.M., and Olsen, B. 2005. Multiple gene segment reassortment between Eurasian and American lineages of influenza A virus (H6N2) in Guillemot (*Uria aalge*). Archives of Virology, 150: 1685–1692.

Wallensten, A., Munster, V. J., Osterhaus, A. D.M. E., Waldenstrom, J., Bonnedahl, J., Broman, T., Fouchier, R. A.M., and Olsen, B. 2006. Mounting evidence for the presence of influenza A virus in the avifauna of the Antarctic region. Antarctic Science, 18: 353–356.

Yoccoz, N. G., Nichols, J. D., and Boulinier, T. 2001. Monitoring biodiversity in space and time: concepts, methods and designs. Trends in Ecology and Evolution, 16: 446–453.

# 7    Climate change and seabirds in the OSPAR Maritime Region

OSPAR has requested ICES to assess the changes in the distribution and abundance of marine species in the OSPAR maritime area in relation to hydrodynamics and sea temperature. WGSE has worked on this request, considering primarily how much evidence of climate-related changes exists for seabirds in the OSPAR Maritime Region.

## 7.1    Overview

Seabirds have long generation times, raise few young per year but live for many years (Ashmole 1971; Jouventin and Mougin, 1981). Because of this life history, populations are able to "integrate" over many consecutive years, so that years in which resources are scarce may take some time to become evident in population trajectories. Also, seabirds seem to be able to sustain episodic, disastrous years, such as occur during El Niño conditions, for example but be more susceptible to more long-term trends in unfavorable conditions (Schreiber, 2002; Veit and Montevecchi, 2006). When searching for responses by seabirds to "climate change", it is important to include changes that occur as long-term trends as well as those that seem to cycle. In the sections that follow, we show how seabirds are impacted by changes in the North Atlantic Oscillation index, which fluctuates periodically, and also how seabirds have responded to more long-term, monotonic change.

Most changes in seabird populations that have been linked to climate are likely mediated through seabird prey. Declines (or increases) are due to changing abundance of prey (Thompson and Ollason, 1981; Veit *et al.,* 1996; Montevecchi and Myers, 1997; Frederiksen *et al.,* 2006), which are in turn driven by environmental change, such as changes in sea temperature. Usually, it has been difficult to establish the link from physical climate forcing through phytoplankton, zooplankton, fish to birds. In the sections that follow we draw what conclusions we feel are reasonable and recognize that some pieces of the puzzle inevitably will be missing.

## 7.2    Changes in breeding distribution

Few seabirds in the OSPAR region have changed their overall *breeding* range in the past 50 years, and for those that have, the role of climate change in such changes is tenuous. Great skuas have increased in abundance within their traditional breeding range in Scotland, the Faeroes and Iceland, and have extended their breeding range north to northern Russia and Svalbard during the last 30 years (Mitchell *et al.,* 2004). Northern Gannets have increased steadily on both sides of the North Atlantic from the late 19th century to the present, and during the last 20 years they have extended their breeding north along the Norwegian coast. In Newfoundland, the major increase during the mid 20th century was related to a shift in diet to mackerel, which moved back into the region when surface waters warmed (Montevecchi and Myers, 1997). Nevertheless, around the North Sea, the largest increase in gannet numbers, during the mid 20th century, did not correspond to the period of greatest temperature change, which is occurring now. Two species of gulls, lesser black-backed gull and Mediterranean gull have expanded their breeding range north during the past 30 years, so in theory these expansions could reflect changing climate. The expansion of lesser black-backed gulls incorporates colonization of, and population growth within, Iceland and greatly increased presence (but not breeding) in North America (Nisbet *et al.*, ms). The expansion of Mediterranean gulls to Britain is part of a broader scale range expansion from the vicinity of the Black Sea westwards. For each species, there are many factors that are influencing these range expansions and it is not clear that climate change is necessarily among these.

## 7.3    Changes in non-breeding distribution

Some seabirds in the OSPAR area have shown changes over recent decades in their migrations and winter distributions. As far as we know, most species have shown very little, or no, change in winter distribution or migration routes. A particular case is the distribution of seabirds that specialize in foraging at the ice-edge. Ivory gulls depend on ice edge habitats for foraging, and their recent steep population decline probably reflects diminishing ice coverage in the Arctic Ocean (Gaston *et al*., 2005).

Common guillemots responded to reductions in sandeel abundance at Shetland in the late 1980s by moving further east in winter to feed in the Skagerrak (Heubeck *et al.,* 1991; Wernham *et al*., 2002).

Northern gannets have increased considerably in <u>breeding</u> numbers in the North Sea, but despite that, the numbers <u>wintering</u> in the North Sea have slightly dropped from the 1980s to 2000s (Garthe *et al.,* in prep). Shipboard transects in the North Sea in winter in the 1980s and early 1990s suggest that virtually as many adult gannets were in the North Sea in winter as in summer. Recent studies suggest that at least half of the gannets present in the North Sea in summer now move out of the North Sea to winter from the Celtic Sea to West Africa. These changes are unlikely to be related directly to climate change, but perhaps rather to food availability and in this case possibly to reductions in the amounts of discards from North Sea fisheries (Kubetzki *et al*., submitted).

Lesser black-backed gulls breeding in the UK have increasingly developed the habit of overwintering in the UK and North Sea rather than migrating to North Africa as they did in previous decades. This progressive change may be linked to milder winter weather and also to available food supplies in the UK (Wernham *et al.,* 2002; Mitchell *et al.,* 2004).

Data on at-sea distribution in the North Sea provides a quantitative basis from which to assess distributional shifts that might reflect climate. Counts of seabirds at sea have been carried out systematically in the North Sea since 1979. These data show that scavengers have declined while a second group of species that includes many non-scavengers have increased (Table 7.1). Large gulls and northern fulmars declined most strongly, matching what would be expected when fisheries effort and thus the availability of discards and offal decline.

**Table 7.1. Overall trends for the two periods (1979–1991 and 1992–2004) covered in the ESAS database 4.1 (Garthe *et al.,* in prep.).**

| OVERALL NORTH SEA TREND | SPECIES |
|---|---|
| **Summer/breeding period** | |
| >50% increase | Northern gannet, lesser black-backed gull, Atlantic puffin |
| 20–50% increase | no species |
| <20% changes | European shag, greater black-backed gull, black-legged kittiwake, common guillemot |
| 20–50% decrease | Northern fulmar, common gull, herring gull |
| >50% decrease | no species |
| **Winter** | |
| >50% increase | Atlantic puffin |
| 20–50% increase | no species |
| <20% changes | European shag, northern gannet, common gull, common guillemot |
| 20–50% decrease | Northern fulmar |
| >50% decrease | Herring gull, greater black-backed gull, black-legged kittiwake |

Trends for the northern and southern North Sea were different. Fulmar, kittiwake and large gull declines were most obvious in the northern half of the North Sea, coinciding with a trend of declining fisheries in that area, while numbers in the southern North Sea did not decline. It thus seems that fisheries effects may have overridden any climatic effects on the at-sea distribution and abundance patterns of seabirds in the North Sea. More detailed investigations are under way to assess these phenomena (Garthe *et al.,* in prep).

## 7.4   Changes in reproductive success

One of the best examples of climate-induced changes in reproductive success to date from the OSPAR region comes from the North Sea. Black-legged kittiwakes have declined by 50% since 1990 (Frederiksen *et al.,* 2004b) and several species experienced breeding failure and/or late breeding in 2004 and 2005 (Wanless *et al.,* 2005, ICES, 2006). The decline in numbers was associated with poor breeding success and lower adult survival over several years (Frederiksen *et al.,* 2004b). The increasing trend in the NAO index to the mid-1990s and the associated warming of the northeastern Atlantic and the North Sea has caused major changes in plankton communities, in particular, a decline in the copepod *Calanus finmarchicus* (Fromentin and Planque, 1996; Planque and Fromentin, 1996). This copepod is often eaten by sandeels, which are in turn a major source of food for several seabird species in the eastern North Atlantic and North Sea (Frederiksen *et al.,* 2006). It is now thought that the bottom-up effect of changing ocean climate conditions causing reductions in forage fish food is a controlling factor in sandeel abundance and quality (Frederiksen *et al.,* 2004b; Wanless *et al.,* 2004; see also Wanless *et al.,* 2005; Frederiksen *et al.,* in press). This is perhaps the best example in the OSPAR region of a link between seabirds and climate through their prey.



Fig. 3. Kittiwake breeding success as a function of local SST in February–March of the previous year and presence/absence of the Wee Bankie sandeel fishery. Data labels indicate current year. Regression lines estimated from weighted multiple regression (see text). Filled circles and solid line, non-fishery years; open symbols and dashed line, fishery years.
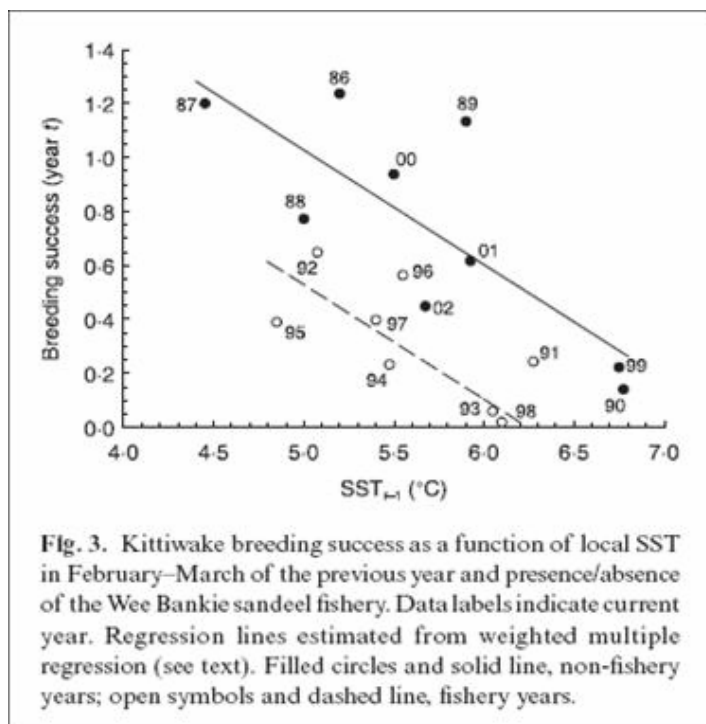
**Figure 7.1. Figure from Frederiksen *et al.* (2004) showing additive effects of the local fishery for sandeels and climate change on kittiwake breeding success on the Isle of May, Firth of Forth, Scotland**

Common and Brünnich's guillemots provide a good example of global-scale population growth in response to climate change (Irons *et al.,* submitted). The two species reacted somewhat differently to SST shifts. The Arctic-adapted Brünnich's guillemot performed best when the SST increased slightly, whereas the more temperate common guillemot did best

when the SST decreased slightly. Both species reacted negatively following the stronger changes in SST (mean SST differing more than 0.5°C from that in the previous regime), regardless of whether the temperatures changes were positive or negative. This response, with the magnitude of the shift being more important than its direction, suggests that the largest shifts were causing the most severe and long-lasting changes to the food webs these birds rely on. This illustrates the complexity of how climate change will impact seabird populations, and emphasizes that extreme care is needed when projecting an observed, short-term trends to the longer-term climate change scenario.

## 7.5    Changes in annual survivorship

The survival of northern fulmars breeding on Eynhallow, Orkney, and common guillemots on Skomer, Wales, was negatively correlated with the winter North Atlantic Oscillation (NAO) index one year previously (Grosbois and Thompson 2005; Votier *et al.,* 2005). The survival of black-legged kittiwakes on the Isle of May, Scotland, declined with winter SST one year previously (Frederiksen *et al.,* 2005). The survival of Atlantic puffins at three colonies in the UK was negatively correlated with summer sea surface temperatures (SST), but at Røst, Norway, the relationship was positive (Harris *et al.,* 2005). At all these colonies, summer SST one year previously affected survival to the following year, except for the Isle of May where it was influenced by SST in the current year (Harris *et al.,* 2005). On Hornøya, Norway, the correlation between survival of four auk species and winter/autumn SST was negative, while that of black-legged kittiwake was positive (Sandvik *et al.,* 2005).

In all of these studies, the authors concluded that climate most likely affected seabird survival via indirect effects on prey availability. For Brünnichs guillemots in Hornøya this was established quantitatively, with survival increasing with the combined abundance of herring and capelin prey which, in turn, declined with SST (Sandvik *et al.,* 2005). For the remainder of studies cited, indirect effects were inferred qualitatively from published literature on relationships between key prey species and climate. Such reasoning explains the contrasting trends of survival with SST for Atlantic puffins in the UK and Røst, since those in the UK feed on sandeels and those in Røst feed on herring, and abundance of these are negatively and positively affected by SST respectively (Toresen and Østvedt, 2000; Arnott and Ruxton, 2002).

The corollary of this conclusion is that links between climate and survival may be inferred from studies where seabird survival has been related to abundance of a prey species known to be sensitive to environmental change. For example, the survival of great skuas, Arctic skuas and black-legged kittiwakes on Shetland varied with the availability of sandeels (Oro and Furness, 2002; Ratcliffe *et al.,* 2002; Davis *et al.,* 2005) and, since sandeel stocks are related to ocean currents and SST (Wright 1996; Arnott and Ruxton, 2002), variability in their survival could ultimately be caused by climatic fluctuations.

## 7.6    Changes in migratory schedule

Among seabirds few examples for changes in migration phenology are available. Data of the Dutch seawatching project proved that little gulls along the Dutch mainland coast passed continuously earlier in spring throughout the last three decades (Figure 7.2). Nowadays, the median numbers are counted almost three weeks earlier than in the seventies. Surprisingly, there is nothing of this kind apparent in any of the tern species (NZG/Club van Zeetrekwaarnemers unpublished data; Camphuysen, C. J., pers. comm.). At Falsterbo, southern Sweden, peak migration of Eurasian wigeons delayed by about 10 days in 1994/2001 compared to the period 1982/1993 (Figure 7.3).
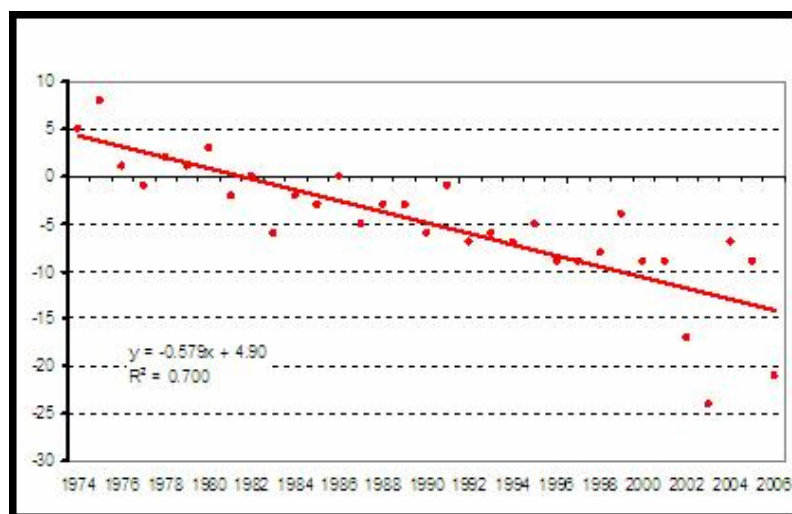
**Figure 7.2. Annual median spring passage times of little gulls at the Dutch mainland coast relative to 30. April (=0). n = 484 385 individuals and 24 734 hours of observation (NZG/Club van Zeetrekwaarnemers unpublished data).**
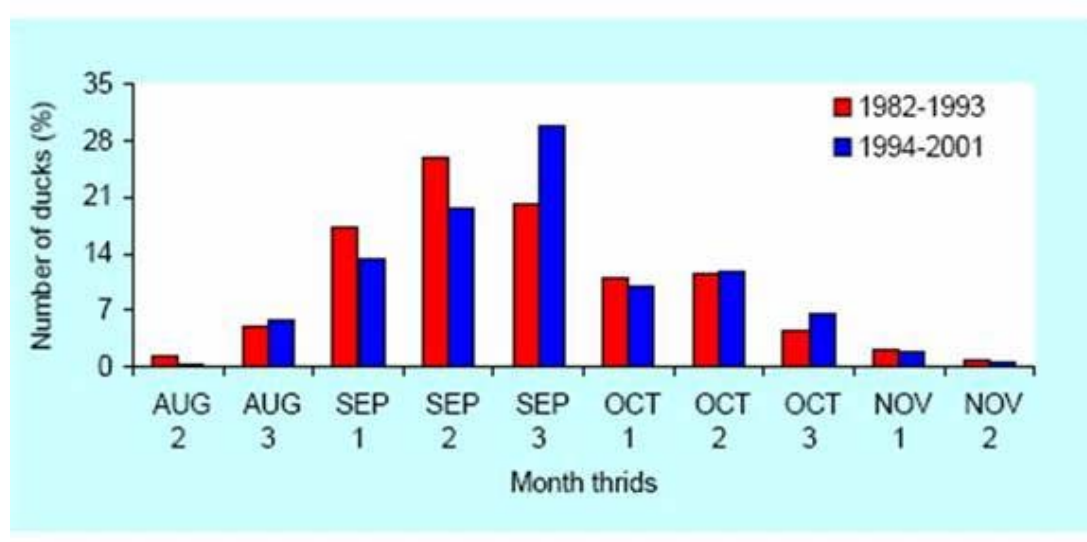


**Figure 7.3. Autumn migration of Eurasian wigeon at Falsterbo, southern Sweden (Bregnballe *et al.,* 2003)**

## 7.7 Other phenological changes: laying dates, nest dates, fledging dates

In general, seabirds commence breeding as early as conditions allow, which for most species in the Northern hemisphere is in April-June. This is driven by a variety of ultimate and proximate factors. It is well known that for their size, seabirds exhibit protracted incubation and chick development periods (Lack, 1968). Thus, an early start to breeding is essential so that chicks are fledged before conditions deteriorate. This adaptation is particularly important in the highly seasonal environments within the northern portions of the OSPAR region, as the summer season is relatively short and winter conditions harsh. Spring is also the time of year when phytoplankton react to high levels of nutrients and increasing amounts of sunlight, and reproduce, creating a rapid increase in primary productivity which eventually results in increased availability of organisms that form the basis of seabird diets (i.e. crustacea, small fishes and squids; Ashmole, 1971). It is generally accepted that birds adjust their timing of

breeding such that the chick rearing period, a time of maximal food and energy requirements for parents, coincides with the seasonal peak in food availability (Lack, 1968).

An ever increasing number of avian studies show that as average temperature increases as a result of climate change, so migration and timing of breeding advance (e.g. Crick, 2004). Among seabirds, there are relatively few studies and the results are equivocal in that timing of breeding is advanced in some cases and retarded in others as a result of changes in climate (Durant *et al.,* 2004a, 2004b; Barbraud and Weimerskirch, 2006). In the North Sea, sea surface temperatures (SSTs) were positively related to egg laying date for Atlantic puffin (Harris *et al.,* 1998) and common guillemot (Harris and Wanless 1988), but negatively related in razorbills (Harris and Wanless, 1989). Late breeding in the face of warmer conditions (the opposite of the general trend in birds) can result from movement or overall decline, in forage species, which have relatively narrow tolerances to temperature due to poikilothermy. This is a frequent occurrence in the Pacific Ocean where thermal perturbations are often forced by ENSOs.

Frederiksen *et al.* (2004) showed that the state of the North Atlantic Oscillation (NAO) index was correlated with timing of breeding for common guillemots and black-legged kittiwakes, both of which disperse in the winter over large spatial scales and thus are in a position to sample large-scale ocean climate variation as indicated by the NAO. Although the NAO is a natural mode of variation in the North Atlantic, it has been suggested by general circulation models of climate that forcing due to human-induced greenhouse gas increases (specifically $CO2$) may cause the NOA index to increase over the next 100 years (Gillet *et al.,* 2003).

In an interesting paper on Atlantic puffins nesting in Røst in the Norwegian Sea, Durant *et al.* (2004b) showed that timing of breeding was negatively related to the NAO index in two periods and not related (or slightly positively related) in an intervening period. They suggested that this was an indication of a regime shift possibly driven by food availability in the year preceding breeding. In a long-term study of Atlantic puffins in the Barents Sea, Barrett (2001) showed that timing of breeding was later in cooler years, with the suggestion that access to nest sites might have been hampered due to ice and snow in years with lower spring temperatures.

In summary then, seabirds appear to react to climate change and variability in a variety of ways. In some circumstances, a warming trend advances timing of breeding and in others breeding is retarded. Seabirds show some flexibility in dealing with climate change in this regard but are ultimately constrained because of the finite (and often lengthy) time required to complete the breeding cycle. Because they are long-lived, seabirds are often able to "buffer" short term (<10 years) environmental variability, especially at the population level. Seabirds are vulnerable to both spatial and temporal mismatches in prey availability, especially when breeding a fixed colony sites with the foraging constraints that these entail (e.g. Weimerskirch *et al.,* 1993).

**Table 7.2. Links between climate change and seabirds.**

| SEABIRD PARAMETER | SPECIES | REGION | CLIMATE VARIABLE | SIGN OF CORRELATION WITH WARMING | SOURCES |
|---|---|---|---|---|---|
| Breeding range | Lesser black-backed gull | Iceland | Sea temperature | Positive | |
| | Lesser black-backed gull | U. K. | Sea temperature | Positive | Mitchell *et al.* 2004 |
| | Northern gannet | U. K. | Sea temperature | Positive | Mitchell *et al.* 2004 |
| Non-breeding range | Lesser black-backed gull | U. K. | | Positive | Wernham *et al.* 2002, Mitchell *et al.* 2004 |
| | Common guillemot | Shetland | Sea temperature, sandeels | | Heubeck *et al.* 1991 |
| Reproductive success | Northern fulmar | Orkney (North Sea) | NAO index | Negative (hatching); positive (fledging) | Thompson and Ollason, 2001 |
| | Atlantic puffin | Røst Norwegian Sea | Sea temperature | Positive | Durant *et al.* 2003 |
| | Atlantic puffin | Røst Norwegian Sea | Salinity | Negative | Durant *et al.* 2006 |
| | Greater black-backed gull | Newfoundland | Sea temperature | Positive | Regehr and Rodway, 1999 |
| | Herring gull | Newfoundland | Sea temperature | Positive | Regehr and Rodway, 1999 |
| | Black-legged kittiwake | Newfoundland | Sea temperature | Positive | Regehr and Rodway, 1999 |
| | Leach's storm-petrel | Newfoundland | Sea temperature | Positive | Regehr and Rodway, 1999 |
| | Black-legged kittiwake | Isle of May (North Sea) | Sea temperature | Negative | Frederiksen *et al.* 2004b |
| Annual survivorship | Northern fulmar | Orkney (North Sea) | NAO index | Negative | Grosbois and Thompson, 2005 |
| | Black-legged kittiwake | Isle of May (North Sea) | Sea temperature | Negative | Frederiksen *et al.* 2004b, 2006 |
| | Atlantic puffin | North Sea, Irish Sea | Sea temperature | Negative | Harris *et al.* 2005 |
| | Atlantic puffin | Røst Norwegian Sea | Sea temperature | Positive | Harris *et al.* 2005 |
| | Atlantic puffin | Norway (Barents Sea) | Sea temperature | Negative | Sandvik *et al.* 2005 |
| | Common guillemot | Norway (Barents Sea) | Sea temperature | Negative | Sandvik *et al.* 2005 |
| | Black-legged kittiwake | Norway (Barents Sea) | Sea temperature | Positive | Sandvik *et al.* 2005 |
| Population change | Common guillemot | Circumpolar | Sea temperature | For both species: | Irons *et al.* submitted |

| SEABIRD PARAMETER | SPECIES | REGION | CLIMATE VARIABLE | SIGN OF CORRELATION WITH WARMING | SOURCES |
|---|---|---|---|---|---|
| | Brünnich's guillemot | Circumpolar | Sea temperature | populations increase with small changes and decrease with large changes | Irons *et al.* submitted |
| | Northern gannet | Newfoundland | Sea temperature | Positive | Montevecchi and Myers, 1997 |
| Nesting (laying or hatching) date | Black-legged kittiwake | Isle of May | NAO index | Positive | Fredericksen *et al.* 2004a |
| | Common guillemot | Isle of May | NAO index | Positive | Fredericksen *et al.* 2004a |
| | Atlantic puffin | St. Kilda | Sea temperature | Positive | Harris *et al.* 1998 |
| | Atlantic puffin | Røst (Norwegian Sea) | NAO winter Index | Negative | Durant *et al.* 2004b |
| | Common guillemot | Isle of May (North Sea) | Sea temperature | Negative | Harris and Wanless, 1988 |
| | Razorbill | Isle of May (North Sea) | Sea temperature | Negative | Harris and Wanless, 1989 |
| | European shag | Isle of May (North Sea) | Wind | Negative | Aebischer and Wanless, 1992 |
| Fledging date | Common guillemot | Baltic Sea | Air temperature | Negative | Hedgren, 1979 |
| Foraging cost | Common guillemot | Isle of May (North Sea) | Stormy weather | Positive | Finney *et al.* 1999 |
| | Northern fulmar | Shetland (North Sea) | Wind speed | Negative | Furness and Bryant, 1996 |

## 7.8 References

Aebischer, N. J., Coulson, J. C., and Colebrook, J. M. 1990. Parallel long-term trends across four marine trophic levels and weather. Nature 347: 753–755.

Aebischer, N., and Wanless, S. 1992. Relationships between colony size, adult non-breeding and environmental conditions for Shags *Phalacrocorax aristotelis* on the Isle of May, Scotland. Bird Study, 32: 43–52.

Ainley, D. G., Sydeman, W. J., and Norton, J. 1995. Upper trophic level predators indicate interannual negative and positive anomalies in the California Current food web. Marine Ecology Progress Series, 118: 69–79.

Ainley, D. G., Spear, L. B., Allen, S. G., and Ribic, C. A. 1996. Temporal and spatial patterns in the diet of the common murre in California waters. Condor, 98: 691–705.

Anderson, P. J., and Piatt, J. F. 1999. Community reorganization in the Gulf of Alaska following ocean climate regime shift. Marine Ecology Progress Series, 189: 117–123.

Anker-Nilssen, T. 1992. Food supply as a determinant of reproduction and population development in Norwegian Puffins *Fratercula arctica*. PhD thesis, University of Trondheim.

Anker-Nilssen, T., and Aarvak, T. 2006. Long-term studies of seabirds in the municipality of Røst, Nordland. Results with focus on 2004 and 2005. Norwegian Institute for Nature research, NINA Report 133.

Arnott, S. A., and Ruxton, G. D. 2002. Sandeel recruitment in the North Sea: demographic, climatic and trophic effects. Marine Ecology Progress Series, 238: 199–210.

Ashmole, N. P. 1971. Sea bird ecology and the marine environment. In: Avian Biology, 1. Farner, D., King, J., and Parkes, K. eds. New York, Academic Press. 1: 224–286.

Barber, R. T., and Chavez, F. P. 1983. Biological consequences of El Nino. Science, 222: 1203–1210.

Barber, R. T., and Chavez, F. P. 1986. Ocean variability in relation to living resources during the 1982–1983 El Nino. Nature, 319: 279–285.

Barbraud, C., and Weimerskirch, H. 2006. Antarctic birds breed later in response to climate change. Proceedings of the National Academy of Science of the U. S., 103: 6248–6251.

Barrett, R. T. 2001. The breeding demography and egg size of Norwegian Atlantic Puffins *Fratercula arctica* and Razorbills *Alca torda* during 20 years of climatic variability. Atlantic Seabirds, 3: 97–112.

Barry, J. P., Baxter, C. H., Sagarin, R. D., and Gilman, S. E. 1995. Climate-related, long-term, faunal changes in a California rocky intertidal community. Science, 267: 672–675.

Beaugrand, G., Brander, K. M., Lindley, A., Souissi, S., and Reid, P. C. 2003. Plankton effect on cod recruitment in the North Sea. Nature, 426: 661–664.

Boersma, P. D. 1977. Breeding patterns of galapagos penguins as an indicator of oceanographic conditions. Science, 200: 1481–1483.

Boersma, P. D. 1998. Population trends of the Galapagos Penguin: impacts of El Nino and La Nina. Condor, 100: 245–253.

Bregnballe, T., Asferg, T., Clausager, I., Noer, H., Clausen, P., and Christensen, T. K. 2003. Vildtbestande, jagt og jagttider i Danmark 2002. En biologisk vurdering af jagtens bæredygtighed som grundlag for jagttidsrevisionen 2003. Danmarks Miljøundersøgelser. http://www.dmu.dk/1_viden/2_Publikationer/3_fagrapporter/rapporter/FR428.pdf

Chastel, O., Weimerskirch, H., and Jouventin, P. 1993. High annual variability in reproductive success and survival of an Antarctic seabird, the snow petrel *Pagodroma nivea*. Oecologia, 94: 278–285.

Chelton, D. B., Bernal, P. A., and McGowan, J. A. 1982. Large-scale interannual physical and biological interaction in the California Current. Journal of Marines Research, 40: 1095–1124.

Christensen, J. H., and Christensen, O. B. 2003. Severe summer flooding in Europe. Nature, 421: 805.

Clark, R. A., Fox, C. J., Viner, D., and Livermore, M. 2003. North Sea cod and climate change- modelling the effects of temperature on population dynamics. Global Change Biology, 9: 1669–1680.

Crick, H. Q.P. 2004. The impact of clmate change on birds. Ibis 146, Supplement 1: 48–56.

Croxall, J. P. 1992. Southern ocean environmental changes: effects on seabird, seal and whale populations. Philosophical Transactions of the Royal Society of London B, 338: 319–328.

Croxall, J. P., Trathan, P. N., and Murphy, E. J. 2002. Environmental change and Antarctic seabird populations. Science, 297: 1510–1514.

Cushing, D. H. 1982. Fisheries and Climate. London: Academic Press.

Davis, S. E., Nager, R. G., and Furness, R. W. 2005. Food availability affects adult survival as well as breeding success of parasitic jaegers. Ecology, 86: 1047–1056.

Divoky, G. 1998. Factors âffecting the growth of a Black Guillemot colony in northern Alaska. PhD Thesis. Alaska: University of Alaska, Fairbanks.

Durant, J. M., Anker-Nilssen, T., and Stenseth, N. C. 2003. Trophic interactions under climate fluctuations: the Atlantic puffin as an example. Proceedings of the Royal Society of London B, 270. 1461–1466.

Durant, J. M., Stenseth, N. C., Anker-Nilssen, T., Harris, M. P., Thompson, P. M., and Wanless, S. 2004a. Marine birds and climate fluctuation in the North Atlantic. In: Marine ecosystems and climate variation–The North Atlantic. A comparative perspective. Stenseth, N. C., Ottersen, G., Hurrell, J. W., and Belgrano, A. eds. Oxford Univ. Press, New York, pp. 95–105.

Durant, J. M., Anker-Nilssen, T., Hjermann, D. Ø., and Stenseth, N. C. 2004b. Regime shifts in the breeding of an Atlantic puffin population. Ecology Letters, 7: 388–394.

Durant, J. M., Anker-Nilssen, T., and Stenseth, N. C. 2006. Ocean climate prior to breeding affects the duration of the nestling period in the Atlantic puffin. Biology Letters, 2: 628–631.

Fiedler, W., Bairlein, F., and Köppen, U. 2004: Using large-scale data from ringed birds fort the investigation of effects of climate change on migrating birds: pitfalls and prospects. In: Advances in Ecological Research 35: Birds and Climate Change. Møller, A. P., Fiedler, W., and Berthold, P. eds. Elsevier Science, London, pp. 49–67.

Erikstad, K., Fauchald, P., Tveraa, T., and Steen, H. 1998. On the cost of reproduction in long-lived birds: the influence of environmental variability. Ecology, 79: 1781–88.

Finney, S., Wanless, S., and Harris, M. 1999. The effect of weather conditions on the feeding behaviour of a diving bird, the Common Guillemot *Uria aalge*. Journal of Avian Biology, 30: 23–30.

Francis, R. C., and Hare, S. R. 1994. Decadal-scale regime shifts in the large marine ecosystems of the northeast Pacific: a case for historical science. Fisheries Oceanography, 3: 279–291.

Francis, R. C., Hare, S. R., Hollowed, A. B., and Wooster, W. S. 1998. Effects of interdecadal climate variability on the oceanic ecosystems ofthe northeast Pacific Ocean. Fisheries Oceanography, 7: 1–21.

Fraser, W. R., Trivelpiece, W. Z., Ainley, D. G., and Trivelpiece, S. G. 1992. Increases in Antarctic penguin populations: reduced competition with whales or a loss of sea ice due to environmental warming? Polar Biology, 11: 525–531.

Frederiksen, M., Harris, M. P., Daunt, F., Rothery, P., and Wanless, S. 2004a. Scale-dependent climate signals drive breeding phenology of three seabird species. Global Change Biology, 10: 1214–1221.

Frederiksen, M., Wanless, S., Harris, M. P., Rothery, P., and Wilson, L. J. 2004b. The role of industrial fishery and oceanographic changes in the decline of the North Sea black-legged kittiwakes. Journal of Applied Ecology, 41: 1129–1139.

Frederiksen, M., Wright, P. J., Harris, M. P., Mavor, R. A., Heubeck, M., and Wanless, S. 2006. Regional patterns of kittiwake *Rissa tridactyla* breeding success are related to variability in sandeel recruitment. Marine Ecology Progress Series, 300: 201- 211.

Frederiksen, M., Furness, R. W., and Wanless, S. In press. Regional variation in the role of bottom-up and top-down processes in controlling sandeel abundance in the North Sea. Marine Ecology Progress Series.

Fromentin, B., and Planque, J.-M. 1996. *Calanus* and environment in the eastern North Atlantic. II. Influence of the North Atlantic Oscillation on *C. finmarchicus* and *C. helgolandicus*. Marine Ecology Progress Series, 134: 111–118.

Furness, R., and Bryant, D. 1996. Effect of wind on field metabolic rates of breeding Northern Fulmars. Ecology, 77: 1181–1188.

Gaston A. J., Gilchrist, H. G., and Hipfner, J. M. 2005. Climate change, ice conditions and reproduction in an Arctic nesting marine bird: Brunnich's guillemot (*Uria lomvia* L.). Journal of Animal Ecology, 74: 832–841.

Gillett, N. P., Graf, H. F., and Osborn, T. J. 2003. Climate change and the North Atlantic Oscillation. Geophysical Monographs, 134: 193–209.

Grosbois, V., and Thompson, P. M. 2005. North Atlantic Climate variation influences survival in adult fulmars. Oikos, 109: 273–290.

Guinet, C., Chastel, O., Koudil, M., Durbec, J. P., and Jouventin, P. 1998. Effects of warm sea-surface temperature anomalies on the blue petrel at the Kerguelen Islands. Proceedings of the Royal Society of London B, 265: 1001–1006.

Hamre, J. 1988. Some aspects of the interrelation between the herring in the Norwegian Sea and the stocks of capelin and cod in the Barents Sea. ICES Doc. C. M. 1988/H:42, 15 pp.

Harris, M., Murray, S., and Wanless´, S. 1998. Long-term changes in breeding performance of Puffins *Fraturcula arctica* on St. Kilda. Bird Study, 45: 371–374.

Harris, M., and Wanless, S. 1988. The breeding biology of guillemots *Uria aalge* on the Isle of May over a six year period. Ibis, 130: 172–192.

Harris, M. P., and Wanless, S. 1989. The breeding biology of Razorbills (*Alca torda*) on the Isle of May. Bird Study, 36: 105–114.

Harris, M. P., and Wanless, S. 1996. Differential responses of Guillemot *Uria aalge* and Shag *Phalacrocorax aristotelis* to a late winter wreck. Bird Study, 43: 220–230.

Harris, M. P., Wanless, S., Murray, S., and Mackley, E. 2005. Isle of May seabird studies in 2004. JNCC Report No.375, Joint Nature Conservation Committee, Peterborough, UK.

Harris, M. P., Anker-Nilssen, T., McCleery, R. H., Erikstad, K. E., Shaw, D. N., and Grosbois, V. 2005. Effects of wintering area and climate on the survival of adult Atlantic puffins *Fratercula arctica* in the eastern Atlantic. Marine Ecology Progress Series, 297: 283–296.

Hayward, T. L. 1997. Pacific Ocean climate change: atmospheric forcing, ocean circulation and ecosystem response. Trends in Ecology and Evolution, 12: 150–154.

Hedgren, S. 1979. Seasonal variation in fledging weight of Guillemots, *Uria aalge*. Ibis, 121: 356–361.

Heubeck, M., Harvey, P. V., and Okill, J. D. 1991. Changes in the Shetland guillemot *Uria aalge* population and the pattern of recoveries of ringed birds, 1959–1990. Seabird, 21: 3–21.

Hughes, L. 2000. Biological consequences of global warming: is the signal already apparent? Trends in Ecology and Evolution, 15: 56–61.

Hüppop, O. 2000. Zum Einfluß von Sturm auf Anzahl, Körperkondition, Sterblichkeit und Ernährung auf Helgoland überwinternder Kormorane (*Phalacrocorax carbo*). Jahresbericht des Instituts für Vogelforschung, 4: 14.

Hurrell, J. W. 1995. Decadal trends in the North Atlantic Oscillation: regional temperatures and precipitation. Science, 269: 676–678.

Hyrenbach, K. D., and Veit, R. R. 2003. Ocean warming and seabird communities of the southern California Current System (1987–1998): response at multiple temporal scales. Deep-Sea Res. II, 50: 2537–2566.

ICES. 2006. Report of the Working Group on Seabird Ecology (WGSE) 2006. ICES CM 2006/LRC:08, Ref. ACE, ACME. 86 pp.

Irons, D., Anker-Nilssen, T., Gaston, A. J., Byrd, G. V., Falk, K., Gilchrist, G., Hario, M., Hjernquist, M., Krasnov, Y. V., Mosbech, A., Olsen, B., Petersen, A., Reid, J., Robertson,

G., Strøm, H., and Wohl, K. D. submitted. Fluctuations in circumpolar seabird populations linked to climate oscillations. Global Change Biology.

Jakobsson, J. 1985. Monitoring and Management of the Northeast Atlantic Herring Stocks. Canadian Journal of Fisheries and Aquatic Sciences, 12, Supplement l: 201–221.

Jouventin, P., and Mougin, J. 1981. Les stratégies adaptatives des oiseaux de mer. Revue Ecologie, Terre et Vie, 35: 217–272.

Kubetzki, U., Garthe, S., Crane, J. E., Mendel, B., and Furness, R. W. Submitted. Changes in gannet migration in relation to altered conditions in the North Sea.

Lack, D. 1968. Ecological adaptations for breeding in birds. Methuen and Co. London.

Lambrechts, M. M., Blondel, J., *et al.* 1997. A single response mechanism is responsible for evolutionary adaptive variation in a bird's laying time. Proceedings of the National Academy of Science of the U. S., 94: 5153–5155.

Lehikoinen, E., Sparks, T. H., and Zalakevicius, M. 2004. Arrival and departure dates. In: Advances in Ecological Research 35: Birds and Climate Change. Møller, A. P., Fiedler, W., and Berthold, P. eds. Elsevier Science, London, pp. 1–31.

Loeb, V., Siegel, V., Holm-Hansen, O., Hewitt, R., Fraser, W., Trivelpiece, W., and Trivelpiece, S. 1997. Effects of sea-ice extent and krill or salp dominance on the Antarctic food web. Nature, 387: 897–900.

Lyver, P. O.B., Moller, H., and Thompson, C. 1999. Changes in Sooty Shearwater *Puffinus griseus* chick production and harvest precede ENSO events. Marine Ecology Progress Series, 188: 237–248.

Marchant, S., and Higgins, P. J. 1990. Handbook of Australian, New Zealand & Antarctic birds. Oxford University Press, Melbourne, Australia.

Mavor, R. A., Pickerell, G., Heubeck, M., and Mitchell, P. I. 2002. Seabird numbers and breeding success in Britain and Ireland, 2001. Peterborough, Joint Nature Conservation Committee. (UK Nature Conservation, No.26.)

Mavor, R. A., Parsons, M., Heubeck, M., Pickerell, G., and Schmitt, S. 2003. Seabird numbers and breeding success in Britain and Ireland, 2002. Peterborough, Joint Nature Conservation Committee. (UK Nature Conservation, No.27.)

Mavor, R. A., Parsons, M., Heubeck, H., and Schmitt, S. 2004. Seabird numbers and breeding success in Britain and Ireland, 2003. Peterborough, Joint Nature Conservation Committee. (UK Nature Conservation, No.28.)

Mavor, R. A., Parsons, M., Heubeck, M., and Schmitt, S. 2005. Seabird numbers and breeding success in Britain and Ireland, 2004. Peterborough, Joint Nature Conservation Committee. (UK Nature Conservation, No.29)

Mavor, R. A., Parsons, M., Heubeck, M., and Schmitt, S. 2006. Seabird numbers and breeding success in Britain and Ireland, 2004. Peterborough, Joint Nature Conservation Committee. (UK Nature Conservation, No.30)

McGowan, J. A., Cayan, D. R., and Dorman, L. M. 1998. Climate-ocean variability and ecosystem response in the northeast Pacific. Science, 281: 210–217.

Mitchell, J. K., and Ericksen, N. J. 1992. Effects of climate change on weather-related disasters. In: Confronting Climate Change; risks, implications and responses. Mintzer, I. M. ed. Cambridge University Press, Cambridge, pp. 141–151.

Mitchell, P. I., Newton, S. F., Ratcliffe, N., and Dunn, T. E. 2004. Seabird Populations of Britain and Ireland. T & A D Poyser; London.

Monaghan, P., Uttley, J. D., and Okill, J. D. 1989. Terns and sandeels: seabirds as indicators of changes in marine fish populations. Journal of Fish Biology 35, 339–340.

Montevecchi, W. A., Garthe, S., and Davoren, G. K. 2006. Biophysical influences on seabird trophic assessments. In: Top predators in marine ecosystems. Boyd, I. L., Wanless, S., and Camphuysen, C. J. eds. Cambridge University Press, UK, pp. 118–130.

Montevecchi, W. A., and Tuck, L. M. 1987. Newfoundland Birds: Exploitation, Study, Conservation. Cambridge, U. S.A: Nuttall Ornithological Club.

Montevecchi, W. A., and Myers, R. A. 1997. Centurial and decadal oceanographic influences on changes in northern gannet populations and diets in the north-west Atlantic: implications for climate change. ICES Journal of Marine Sciences, 54: 608–614.

Montevecchi, W. A., and Berruti, A. 1991. Avian indication of pelagic fishery conditions in the Southeast and Northwest Atlantic. Acta XX Congressus Internationalis Ornithologici: 2246–2256.

Morrison, M. L. 1986. Bird populations as indicators of environmental change. In: Johnston RF ed. Current Ornithology. New York: Plenum Press, 429–451.

Newton, S. 2004. Roseate Tern *Sterna dougallii*. In: Seabird Populations of Britain and Ireland. Mitchell, P. I., Newton, S. F., Ratcliffe, N., and Dunn, T. E. T. & Poyser, A. D., London, pp. 302–314.

Nisbet, I. C.T., Veit, R. R., and Auer, S. A. ms. Status of seabirds of the Atlantic coast of the United States. Nuttall Publication Series.

Oro, D., and Furness, R. W. 2002. Influences of food availability and predation on survival of Kittiwakes. Ecology, 83: 2516–2528.

Pedersen, S. A., Lewy, P., Wright, P. 1999. Assessments of the lesser sandeel (*Ammodytes marinus*) in the North Sea based on revised stock divisions. Fisheries Research, 41: 221–241.

Piatt, J. F., and Anderson, P. 1996. Response of common murres to the Exxon Valdez oil spill and long-term changes in the Gulf of Alaska marine ecosystem. American Fisheries Society Symposium, 18: 720–737.

Pickerell, G. 2004. Little Tern *Sterna albifrons*. In: Seabird Populations of Britain and Ireland. Mitchell, P. I., Newton, S. F., Ratcliffe, N., and Dunn, T. E. T. & Poyser, A. D., London, pp. 339–349.

Planque, J.-M., and Fromentin, B. 1996. *Calanus* and environment in the eastern North Atlantic. I. Spatial and temporal patterns of *C. finmarchicus* and *C. helgolandicus*. Marine Ecology Progress Series, 134: 101–109.

Proctor, R., Wright, P. J., and Everitt, A. 1998. Modelling the transport of larval sandeels on the north west European shelf. Fisheries Oceanography, 7: 347–354.

Ratcliffe, N., Catry, P., Hamer, K. C., Klomp, N. I., and Furness, R. W. 2002. The effects of age and year on the survival of breeding great skuas *Catharacta skua* in Shetland. Ibis, 144: 384–392.

Regehr, H. M., and Montevecchi, W. A. 1997. Interactive effects of food shortage and predation on breeding failure of black-legged kittiwakes: indirect effects of fisheries activities and implications for indicator species. Marine Ecology Progress Series, 155: 249–260.

Regehr, H., and Rodway, M. 1999. Seabird breeding performance during two years of delayed Capelin arrival in the Northwest Atlantic: a multi-species comparison. Waterbirds, 22: 60–67.

Rindorf, A., Wanless, S., and Harris, M. P. 2000. Effects of changes in sandeel availability on the reproductive output of seabirds. Marine Ecology Progress Series, 202: 241–252.

Roemmich, D. 1992. Ocean warming and sea level rise along the southwest U. S. coast. Science, 257: 373–375.

Roemmich, D., and McGowan, J. A. 1995. Climatic warming and the decline of zooplankton in the California Current. Science, 267: 1324–1326.

Sandvik, H., Erikstad, K. E., Barrett, R. T., and Yoccoz, N. G. 2005. The effect of climate on adult survival in five species of North Atlantic seabirds. Journal of Animal Ecology, 74: 817–831.

Scheffer, M., Carpenter, S., Foley, J. A., Folke, C., and Walker, B. 2001. Catastrophic shifts in ecosystems. Nature, 413: 591–596.

Schreiber, R. W., and Schreiber, E. A. 1984. Central pacific seabirds and the El Nino-Southern Oscillation: 1982–1983 perspectives. Science, 225: 713–716.

Schreiber, E. A. 2002. Climate and weather effects on seabirds. In: Biology of Marine Birds. Schreiber, E. A., and Burger, J. ed. Boca Raton, Louisiana, CRC Press, pp. 179–215.

Siriwardena, G. M., and Wernham, C. V. 2002. Synthesis of the migration patterns of British & Irish birds. In: The Migration Atlas: movements of the birds of Britain and Ireland. Wernham, C. V., Toms, M. P., Marchant, J. H., Clark, J. H., Siriwardena, G. M., and Baillie, S. R. eds. T. & Poyser, A. D., London, pp. 70–102.

Smith, R. C., Ainley, D., Baker, K., Domack, E., Emslie, S., Fraser, B., Kennett, J., Leventer, A., Mosley-Thompson, E., Stammerjohn, S., and Vernet, M. 1999. Marine ecosystem sensitivity to climate change. BioScience, 49: 393–404.

Springer, A. M. 1998. Is it all climate change? Why marine bird and mammal populations fluctuate in the North Pacific. In: Biotic Impacts of Extratropical Climate Variability in the Pacific. Holloway, G., Muller, P., Henderson, D. ed. SOEST Special Publication, NOAA Award No. NA67RJ0154.

Springer, A. M., Roseneau, D. G., Murphy, E. C., and Springer, M. I. 1984. Environmental controls of marine food webs: food habits of seabirds in the eastern Chukchi Sea. Canadian Journal of Fisheries and Aquatic Sciences, 41: 1202–1215.

Springer, A. M., Roseneau, D. G., Lloyd, D. S., McRoy, C. P., and Murphy, E. C. 1986. Seabird responses to fluctuating prey availability in the eastern Bering Sea. Marine Ecology Progress Series, 32: 1–12.

Springer, A. M., Murphy, E. C., Roseneau, D. G., McRoy, C. P., and Cooper, B. A. 1987. The paradox of pelagic food webs in the northern Bering Sea — I. Seabird food habits. Continental Shelf Research, 7: 895–911.

Steele, J. H. 1998. Regime shifts in marine ecosystems. Ecological Applications, 8: S33–S36.

Thompson, P., and Ollason, J. C. 2001. Lagged effects of ocean climate change on fulmar population dynamics. Nature, 413: 417–420.

Thompson, P. M., and Grosbois, V. 2002. Effects of climate variation on seabird population dynamics. Directions in Science, 1: 50–52.

Toresen, R., and Østvedt, O. J. 2000. Variation in abundance of Norwegian spring-spawning herring (*Clupea harengus*, Clupeidae) throughout the 20 th century and the influence of climatic fluctuations. Fish and Fisheries, 1: 231–256.

Uttley,J., Monaghan, P., and White, S. 1989. Differential effects of reduced sandeel availability on two sympatrically breeding species of tern. Ornis Scandinavica, 20: 273–277.

Veit, R. R., Pyle, P., and McGowan, J. A. 1996. Ocean warming and long-term change in pelagic bird abundance within the California Current System. Marine Ecology Progress Series, 139: 11–18.

Veit, R. R., McGowan, J. A., Ainley, D. G., Wahl, T. R., and Pyle, P. 1997. Apex marine predator declines 90% in association with changing oceanic climate. Global Change Biology, 3: 23–28.

Veit, R. R., and Montevecchi, W. A. 2006. The influences of climate change on marine birds. Acta Zoologica Sinica 52, Supplement: 165–168.

Votier, S. C., Hatchwell, B. J., Beckerman, A., McCleery, R. H., Hunter, F. M., Pellatt, J., Trinder, M., and Birkhead, T. R. 2005. Oil pollution and climate effects have wide-scale impacts on seabird demographics. Ecology Letters, 8: 1157–1164.

Wanless, S., Wright, P. J., Harris, M. P., and Elston, D. A. 2004. Evidence for decrease in size of lesser sandeels *Ammodytes marinus* in a North Sea aggregation over a 30-yr period. Marine Ecology Progress Series, 279: 237–246.

Wanless, S., Harris, M. P., Redman, P., and Speakman, J. R. 2005. Low energy values of fish as a probable cause of a major seabird breeding failure in the North Sea. Marine Ecology Progress Series, 294: 1–8.

Weimerskirch, H. 2001. Seabird demography and its relationship with the marine environment. In: Biology of Marine Birds. Schreiber, E. A., and Burger, J. ed. Boca Raton, Louisiana, CRC Press, pp. 115–135.

Weimerskirch H, Salamolard, M., Sarrazin, F., and Jouventin, P. 1993. Foraging strategy of wandering albatrosses through the breeding season: A study using satellite telemetry. Auk, 110: 325–342.

Weimerskirch, H., Barbraud, C., and Lys, P. 2000. Sex differences in parental investment and chick growth in Wandering Albatrosses: fitness consequences. Ecology, 81: 309–318.

Wernham, C. V., Toms, P. M., Marchant, J. H., Clark, J. A., Siriwardena, G. M., and Baillie, S. R. eds. 2002. The Migration Atlas: Movements of the Birds of Britain & Ireland. T & AD Poyser, London.

Williams, G. 1966. Natural selection, the costs of reproduction, and a refinement of Lack's principle. American Naturalist, 100: 687–690.

Wiltshire, K. H., and Manly, B. F.J. 2004. The warming trend at Helgoland Roads, North Sea: phytoplankton response. Helgoland Marine Research, 58: 269–273.

Wright, P. J. 1996. Is there a conflict between sandeel fisheries and seabirds? A case study at Shetland. In: Aquatic predators and their prey. Greenstreet, S. P.R., and Tasker, M. L. eds. Fishing News Books, Oxford, pp. 154–165.

Wright, P. J., Jensen, H., and Tuck, I. 2000. The influence of sediment type on the distribution of the lesser sandeel, *Ammodytes marinus*. Journal of Sea Research, 44: 243–256.

## 8 Review of OSPAR seabird nominations for threatened and/or declining species and habitats

This chapter has been delivered to ICES ACE during the WGSE meeting as part of the OSPAR request.

### 8.1 Overview

#### 8.1.1 Inadequate Time

ICES WGSE appreciates the activities of OSPAR to review and enlarge the list of threatened and/or declining species. However, due to the WGSE structure and participants' schedules, it was not possible to fully respond to this request by the given deadline in January 2007. The Working Group on Seabird Ecology only meets once per year, usually in spring. The WGSE group met today (20 March 2007) and managed to finalise comments on all nine species listed in the OSPAR request. The comments are summarised and specified in this document.

ICES advice has been based on our preliminary assessment of the five species as provided in January 2007. This version now contains updated information on the five species that does not conflict with the preliminary assessment.

**Table 8.1. Summary of reviewed species nominations.**

| Species | Accept | Accept with minor revisions | Reconsider with Major Revisions | Data deficiency to assess status | Reject because OSPAR is a peripheral area for this species only |
|---|---|---|---|---|---|
| *Gavia arctica* (Arctic Loon = Black-throated Diver) | | | | | X |
| *Oceanodroma castro* (Band-rumped Storm-Petrel) | | X | | | |
| *Puffinus mauretanicus* (Balearic Shearwater) | | X | | | |
| *Phalacrocorax aristotelis* (European Shag) | | | X | | |
| *Aythya marila* (Greater Scaup) | | | | X | |
| *Melanitta fusca* (White-winged Scoter = Velvet Scoter) | | | | | X |
| *Pagophila eburnea* (Ivory Gull) | | X | | | |
| *Rissa tridactyla* (Black-legged Kittiwake) | | | X | | |
| *Uria lomvia* (Thick-billed Murre = Brünnich's Guillemot) | | X | | | |

#### 8.1.2 References contained within the nine seabird nominations

WGSE acknowledges that there are several indications in the published literature that the nine nominated species have declined in numbers and/or are threatened. However, it appears that the nomination documents have not yet fully acknowledged important references available documenting these species declines and/or threats. Whereas publications by Birdlife

International are consistently quoted, other more specialised references are often not mentioned. The comments, below, by WGSE provide some examples of references that would be useful to add to those species nominations.

## 8.2 Species Reviews

In the forthcoming sections, the following nine species will be reviewed:

1 ) *Gavia arctica* (Arctic loon)
2 ) *Oceanodroma castro* (Band-rumped storm-petrel)
3 ) *Puffinus mauretanicus* (Balearic shearwater)
4 ) Phalacrocorax aristotelis (European shag)
5 ) *Aythya marila* (Greater scaup)
6 ) *Melanitta fusca* (White-winged scoter)
7 ) *Pagophila eburnea* (Ivory gull)
8 ) *Rissa tridactyla* (Black-legged kittiwake)
9 ) *Uria lomvia* (Thick-billed murre)

### 8.2.1   Arctic Loon = Black-throated Diver (*Gavia arctica*)

*Known distribution*

The general autumn migration direction of Scandinavian black-throated divers is south to southeast, with main wintering grounds in the Black Sea, Mediterranean Sea and Caspian Sea. The UK breeding population is estimated to be less than 200 pairs (http://blx1.bto.org/birdfacts/results/bob30.htm) and although many gather off the west coast of Scotland in spring, their precise wintering grounds are unknown. The winter population of the species in the OSPAR area is estimated at 700 individuals.

*Sufficiency of data*

The BirdLife note on the status of black-throated diver describe the North Sea off German and Danish Waddenzee as well as northern Kattegat in Denmark to be the most important sites within the OSPAR region. These data most likely derive from data collected 15–20 years ago. More recent data collected in both German and Danish waters suggest that the vast majority of divers present in this area were red-throated divers *G. stellata* (Garthe 2003, Petersen *et al.,* 2006). Data collected by aerial surveys have a high proportion of recordings unidentified *G. stellata/arctica*. Even then the data show that the majority of divers are *G. stellata* in the North Sea and in Kattegat. This pattern was supported by data from migration studies on Helgoland (Dierschke 2002), where 1% of the passing divers were *G. arctica*, and Skagen (Rolf Christensen, in litt.).

Thus, the wintering population of *G. arctica* in the OSPAR region at present seems to be of an insignificant size when compared to the population as a whole. The reason for this change is difficult to evaluate. Change in wintering quarters is a possibility, but potentially difficulties with species identification between *G. stellata* and *G. arctica* in previous times could play a role.

*Threats*

Offshore wind farms are mentioned by BirdLife International as a threat, referring to Christensen and Houinsen (2005). In the area of the wind farm *G. stellata* was the dominant species (53% of confirmed sightings) while less than 2% were identified as *G. arctica* (the remainder of observed divers were unidentified, Petersen, 2005). Thus, the response from the wind farm on divers will be based almost entirely on observations of *G. stellata*. There is on

the other hand no reason to suggest that the two diver species would react differently on wind turbines, but given the low frequency of occurrence of the Black-throated Diver in the vicinity of the wind farms, mentioning the fact appears appropriate.

*Useful references*

Christensen, T. K., and Hounisen, J. P. 2005. Investigations of migratory birds during operation of Horns Rev offshore wind farm: Preliminary note of analysis of data from spring 2004. Department of Wildlife Ecology and Biodiversity, National Environmental Research Institute, Ministry of the Environment, Denmark. NERI Note commissioned by Elsam Engineering A/S 2005.

Dierschke, V. 2002. Migration of Red-throated Divers *Gavia stellata* and Black-throated Divers *G. arctica* near Helgoland (SE North Sea). Vogelwelt, 123: 203–211.

Garthe, S. 2003. Verteilungsmuster und Bestände von Seevögeln in der Ausschließlichen Wirtschaftszone (AWZ) der deutschen Nord- und Ostsee und Fachvorschläge für EU-Vogelschutzgebiete. Berichte zum Vogelschutz, 40: 15–56.

Petersen, I. K. 2005. Bird numbers and distributions in the Horns Rev offshore wind farm area. Annual status report 2004. NERI Report Commissioned by Elsam Engineering A/S.

Petersen, I. K., Christensen, T. K., Kahlert, J., Desholm, M., and Fox, A. D. 2006. Final results of bird studies at the offshore wind farms at Nysted and Horns Rev, Denmark, NERI Report. Commissioned by DONG Energy and Vattenfall A/S 2006.

### 8.2.2  Band-rumped Storm-Petrel (*Oceanodroma castro*)

*Evidence for sensitivity*

There is no indication where the estimate of generation length comes from. Bolton *et al.* (2004) may be cited as evidence for inter-specific competition for nest burrows in Azorean colonies where breeding is confirmed.

*Global importance*

It is stated that the "Monteiro" population is confined to the islets in Azores. There is evidence that the hot-season breeding population in Desertas islands, Madeira archipelago is distinct from that population (Friesen *et al.,* in press). Genetic analysis strongly supports the separation of the Azores 'hot season' form as a separate taxon.

*Required further management*

Further investigation on the comparison of the "Monteiro" population from the hot-season breeding population in Desertas islands, Madeira archipelago (the nearest hot-season breeding population in the Atlantic) is needed. The results from that will determine the total abundance of this population and will influence the level of threat and the future management required.

*Useful references*

Add the following references:

Bolton, M., Medeiros, R., Hothersall, B., and Campos, A. 2004. The use of artificial breeding chambers as a conservation measure for cavity-nesting procellariiform seabirds: a case study of the Madeiran storm petrel (*Oceanodroma castro*). Biological Conservation, 116: 73–80.

Friesen, V. L., *et al.* in press. Sympatric speciation by separation in breeding time in a seabird. Proceedings of the National Academy of Science of the U. S.

### 8.2.3    Balearic Shearwater (*Puffinus mauretanicus*)

*Known distribution and annual cycle*

There are a number of inaccuracies regarding the annual cycle of the Balearic shearwater. Basically, there is constant confusion between post-breeding movements and winter distribution. The species leaves the Mediterranean after breeding, in early July (non-breeders start moving in May), when it can be found from northern Portugal to the UK, coinciding with spawning of small pelagic fish in these areas (see Le Mao and Yésou 1993 for details). Birds start returning to the Mediterranean in September, with the major influx in October-December, although some individuals can remain in the Atlantic throughout the winter. Also a few birds can remain in the Mediterranean and Alboran seas during the post-breeding time. Thus, the relevance of the OSPAR area for the species is mainly during the post-breeding period (not winter).

Concerning foraging ecology, also mention the use of trawling discards, which provide over 40% of the energy requirements of the species during the breeding period (Arcos and Oro 2002). The species tend to concentrate in productive areas characterized by oceanographic fronts, at least during the breeding period (Louzao *et al.,* 2006).

*Global importance*

No doubt the breeding population is small and requires urgent conservation action. However, this should be stressed that counting the breeding population of such a seabird is a difficult task, and that efforts should be directed towards a comprehensive census. Population trends based on population estimates throughout time should be taken with care, as sometimes the outcome of a census depends on the methodology employed and the effort invested, and strong biases could confound actual population trends. This is important to use focal colonies to assess population trends with some accuracy.

*Evidence for sensitivity*

There is no indication where the estimate of generation length comes from. The only published information comes from Oro *et al.* (2004).

The tendency of forming large flocks in coastal areas (up to 5000 individuals) should be mentioned in support of the sensitivity of the species, particularly when facing oil-spills. This can also have implications for the by-catch in long-liners or other fishing gears, as the entanglement of birds could be infrequent but numerically important when it happens close to a raft (e.g. cases of up to 50 birds caught in a single line).

*Evidence for decline*

The most reliable information supporting a population decline comes from demographic studies (Oro *et al.,* 2004). It is necessary to extend these studies as to include a major number of colonies. Trends based on global population estimates throughout time are less reliable (see *Global importance*).

*Sufficiency of data*

See above considerations concerning population estimates. Evidence in the Mediterranean concerning threats is not as good as suggested. Ruiz and Marti (2004) point out the main threats.

*Management considerations*

Maybe it'd be worth mentioning the efforts of SEO/BirdLife and SPEA to create an inventory of marine IBAs in Spain and Portugal, respectively, as a first step towards the effective

protection of the foraging grounds off the Iberian Peninsula (including the OSPAR area) and the Balearics.

Reference to inshore concentrations "when roosting" should be reconsidered, as these concentrations may partly be for feeding purposes (in areas of high density of small pelagic fish).

*Minor comments*

- Maybe mention that *P. mauretanicus* was first treated as a subspecies of *P. puffinus*, and later of *P. yelkouan* (in agreement with Heath and Evans, 2000).
- Change Sauleda 2005 to Viada 2005 (correct in the reference list).

*Useful references*

Add/modify the following references:

Arcos, J. M., and Oro, D. 2002. Significance of fisheries discards for a threatened Mediterranean seabird, the Balearic shearwater Puffinus mauretanicus. Marine Ecology Progress Series, 239: 209–220.

Arcos, J. M., Louzao, M., and Oro, D. in press. Fishery Ecosystem Impacts and Management in the Mediterranean: Seabirds Point of View. *In*: J. Nielsen, J. Dodson, K. Friedland, T. Hamon, N. Hughes, J. Musick, and E. Verspoor (Eds). Proceedings of the Fourth World Fisheries Congress: Reconciling Fisheries with Conservation. American Fisheries Society, Symposium 49, Bethesda, Maryland, in press.

Estrategia para la conservación de la Pardela balear *Puffinus mauretanicus* en España (2005). Elaborada por el Grupo de Trabajo de la Pardela balear. Ministerio de Medio Ambiente.

Le Mao P., and Yésou, P. 1993. The annual cycle of Balearic shearwaters and western Mediterranean Yellow-legged Gulls: some ecological considerations. In: Aguilar J.S., Monbailliu X., Paterson, A.M. (eds) Status and conservation of seabirds. Proceedings of the 2nd Mediterranean Seabird Symposium, SEO/BirdLife, Madrid, p 135–145.

Louzao, M., Hyrenbach, D., Arcos, J. M., Abelló, P., Gil de Sola, L., and Oro, D. 2006. Oceanographic habitat of a critically endangered Mediterranean Procellariiform: implications for the design of Marine Protected Areas. Ecological Applications, 16: 1683–1695.

Louzao, M., Igual, J. M., McMinn, M., Aguilar, J. A., Triay, R., and Oro, D. 2006. Small pelagic fish, trawling discards and breeding performance of the critically endangered Balearic Shearwater: improving conservation diagnosis. Marine Ecology Progress Series, 318: 247–254.

Mouriño,J., Arcos, F., Salvadores, R., Sandoval, A., and Vidal, C. 2003. Status of the Balearic shearwater (*Puffinus mauretanicus*) on the Galician coast (NW Iberian Peninsula). Scientia Marina 67, pp. 135–142

Ruiz, A., Marti R. eds. 2004. La Pardela Balear. SEO/Birdlife y Govern de Les Illes Balears. 220 pp.

### 8.2.4 European Shag (*Phalacrocorax aristotelis*)

*Evidence for sensitivity*

In addition to the Spanish studies quoted, several UK studies also demonstrate the sensitivity of shags to e.g. winter weather: Aebischer and Wanless (1992), Aebischer (1993), Harris and Wanless (1996), Harris *et al.* (1998). The paper quoted by Wanless *et al.* (2005) does not claim that shags are sensitive to declines in sandeel quality (but does document extremely low quality of sandeels in 2004).

*Evidence for threats*

The statements attributed to Mitchell *et al.* (2004) actually come from Rindorf *et al.* (2000). In addition, Frederiksen *et al.* (2006) showed a positive relationship between shag breeding productivity and the abundance of sandeel larvae in the previous year. The papers by Aebischer and Wanless (1992) and Aebischer (1993) is also relevant for the discussion of how future weather may affect shags. Almost one third of the world population breeds in Norway where numbers have been increasing by approximately 10% p.a. in most areas over the last decade (Barrett *et al.,* in press).

*Required further management*

The conclusion that sandeels are the main prey of shags everywhere is premature. While sandeels are very important e.g. in the North Sea, shags are opportunistic predators and probably rely on other fish species in other areas. Shag diet has been studied in very few locations, and there is a clear need for more studies of this type, before strong conclusions are drawn about links between shag declines and low sandeel recruitment. Gadids are probably among the most important prey species in the Norwegian and Barents Seas. Regarding disturbance, shags can show a remarkable tolerance to human presence, as long as visitor access and activities are strictly regulated (e.g. on the Farne Islands).

*Useful references*

Change reference to Mitchell *et al.* (2004) to chapter reference (Wanless and Harris 2004).

Add the following references (as a minimum):

Aebischer, N. J., and Wanless, S. 1992. Relationships between colony size, adult non-breeding and environmental conditions for Shags *Phalacrocorax aristotelis* on the Isle of May, Scotland. Bird Study, 39: 43–52.

Aebischer, N. J. 1993. Immediate and delayed effects of a gale in late spring on the breeding of the Shag *Phalacrocorax aristotelis*. Ibis, 135: 225–232.

Barrett, R. T., Lorentsen, S.-H., and Anker-Nilssen, T. in press. The status of breeding seabirds in mainland Norway. Altantic Seabirds.

Frederiksen, M., Edwards, M., Richardson, A. J., Halliday, N. C., and Wanless, S. 2006. From plankton to top predators: bottom-up control of a marine food web across four trophic levels. Journal of Animal Ecology, 75: 1259–1268.

Harris, M. P., and Wanless, S. 1996. Differential responses of Guillemot *Uria aalge* and Shag *Phalacrocorax aristotelis* to a late winter wreck. Bird Study, 43: 220–230.

Harris, M. P., Wanless, S., and Elston, D. A. 1998. Age-related effects of a non-breeding event and a winter wreck on the survival of Shags *Phalacrocorax aristotelis*. Ibis, 140: 310–314.

Rindorf, A., Wanless, S., and Harris, M. P. 2000. Effects of sandeel availability on the reproductive output of seabirds. Marine Ecology Progress Series, 202: 241–252.

Wanless, S., and Harris, M. P. 1997. *Phalacrocorax aristotelis* Shag. BWP Update, 1: 3–13.

Wanless, S., and Harris, M. P. 2004. European Shag *Phalacrocorax aristotelis*. Pages 146–159 *in* Mitchell, P. I., Newton, S. F., N. Ratcliffe, and Dunn, T. E., editors. Seabird populations of Britain and Ireland. T. & Poyser, A. D., London.

### 8.2.5   Greater Scaup (*Aythya marila*)

The population size of wintering scaup in NW Europe was estimated in 1994 based on comprehensive surveys in Northwest Europe particularly those in the Baltic Sea in 1992 and 1993.

Since then no simultaneous, comprehensive surveys have been carried out in the area (although such surveys were planned for the winter 2007), making it difficult to provide any effective assessment of true changes in population status and/or distribution since that time.

Annual surveys have, however been carried in the Netherlands and Germany. These surveys show a marked decrease in numbers from the middle of the 1990s. In Denmark a similar decrease seems to have taken place although figures only exist from 2000. In the fourth country, Poland, where large numbers of scaup were recorded in the beginning of the 1990s no comprehensive data exists after 1993.

The conclusions in the BirdLife document on scaup and in the EU Management Plan for Scaup are thus based on insufficient data. The status of breeding scaup in Scandinavia might be well documented, but the status of the vast majority of scaup which breed in Russia remains unknown. In lake Mývatn, Iceland, the breeding population has been monitored annually since 1974. During this period the population has been remarkably stable in this important breeding site, with approximately 1.000 to 2.000 males. Regarding wintering site counts, figures from Northern Europe are probably sufficient, but with the notable exception of Poland.

It is very important to obtain more recent figures for the wintering stock of scaup in Poland. In 1993 9300 Scaup were recorded in Poland but in the previous years 25 500 and 42 200 scaup were counted. Given that the January 0° isotherm is increasingly moving into the Baltic Sea, the western parts of Poland's marine waters are now more often ice free in winter, especially those of the large coastal lagoons which provide much better feeding conditions for scaup than in previous years. It seems more likely that considerable numbers of scaup winter here, with consequent reductions in numbers passing further west to resorts further down the flyway into Germany and the Netherlands.

A reasonable conclusion would be that the scaup may be declining and/or suffering a major winter redistribution, but the magnitude of any real decline is not possible to document.

The BirdLife paper states that the reasons for the declines are unknown, but mentions some factors that might be relevant, including hunting. Denmark is one of the few European countries where the scaup is legal quarry. The estimated bag has decreased over recent years and the current Danish bag is estimated to be only a few hundred birds. Altogether the total EU bag is probably less than 1000 birds. It seems thus highly unlikely that hunting mortality (but also including any effects of hunting disturbance) could contribute to being a major cause of the decline. The catch of mussel and cockles in the Netherlands including the *Spisula* fishing is also mentioned. However, the scaup in the Netherlands are known to mostly winter in the IJsselmeer, where they feed mainly on zebra mussels.

Drowning in fishing gear might be the biggest threat to scaup in the wintering quarters in NW Europe, but its true magnitude is unknown. Even large numbers of birds might die from this source of mortality, it is not sufficient to explain any marked decrease in the species numbers.

Looking at the situation over the last 20 years the conditions for wintering scaup in NW Europe seem to have improved. In general, the hunting pressure has decreased, the feeding conditions (with the possible exception of the Netherlands) seem to have improved and oil spills have decreased both in frequency and severity.

If there has been a real decline in overall numbers, the main reason for the decrease is therefore unlikely to be found on the wintering quarters. It therefore remains a possibility that declines could be related conditions on the breeding areas, either the result of deterioration of suitable breeding habitat or problems associated with the population maintaining sufficient reproduction.

Information about the breeding success in scaup and the status and condition of the breeding habitats of the species is scarce or nonexistent.

Due to the major lack of knowledge relating to scaup, the uncertainty relating to its precise distribution and abundance in relation to the earlier results from the 1990s and the reasons for the apparent decrease in numbers, conservation actions associated with this species should in the first instance concentrate on obtaining the necessary information to document a decline and then to determine the reasons for the decrease. On that basis, active conservation measures can be taken either to improve and conserve the wintering quarters or – more likely – the breeding habitats.

### 8.2.6   White-winged Scoter = Velvet Scoter (*Melanitta fusca*)

The vast majority of White-winged Scoters winter in the Baltic Sea and thus outside the OSPAR area.

The population size of wintering white-winged scoter in NW Europe was estimated in 1994 based on comprehensive surveys in Northwest Europe in 1992 and 1993. Since then no simultaneous, comprehensive surveys have been carried out in the area (although such surveys were planned for the winter 2007), making it difficult to provide any effective assessment of true changes in population status and/or distribution since that time.

Annual surveys have been carried out in some of the NW European countries, but only France and UK regularly holds a few thousand birds or more, and the total number of White-winged Scoters outside the Baltic Sea amount to less than 10.000.

It is possible that current data from the OSPAR area suggests a decline in white-winged scoter, but as this area only holds 1–2% of the total population it is doubtful if any generic trends can be obtained from this area.

It probably is less important to look at factors for any possible decrease. The commercial harvest of mussel and cockles in the Netherlands including the *Spisula* fishing is mentioned as a reason, but with a winter total of ~50 birds in the Netherlands this is insignificant.

### 8.2.7   Ivory Gull (*Pagophila eburnea*)

*Known distribution*

In Canada, colonies are often found on rocky promontories surrounded by ice fields. These are called nunataks.

*Evidence for sensitivity*

The species appears to be sensitive to overflights by aircraft. It is agreed that the species is "very sensitive".

*Evidence for decline*

Canadian studies clearly show that ivory gull populations in that country have declined dramatically over the past 20 or more years (see references below). Mention of this would add to and support the statements made about declines in the OSPAR region.

*Evidence for threats*

Recent work in Canada has shown that Canadian ivory gull eggs have exceptionally high levels of mercury and it is thought that this may be an important factor in the observed population decline (G. Gilchrist pers. comm.). Although it is agreed that climate change and warming of the Arctic is a significant long-term threat to this species mercury contamination may be ranked above this, at least in Canada, and in the shorter term. The species is currently listed as Endangered by the Committee on Endangered Species in Canada (COSEWIC) and is being considered for listing under the Canadian Species at Risk Act.

The case for including the ivory gull in the OSPAR list may be strengthened by mentioning the 'Conservation of Arctic Flora and Fauna' (CAFF) CBIRD 'Draft Conservation Strategy of the Ivory Gull' (unpubl. report).

*Useful references*

Chardine, J. W., Fontaine, A. J., Blokpoel, H., Mallory, M., and Hofmann, T. 2004. At-sea observations of Ivory Gulls (*Pagophila eburnea*) in the eastern Canadian High Arctic I 1993 and 2002: Indication of a population decline. Polar Record, 40: 355–359.

Frisch, T., 1983. Ivory gull colonies on the Devon Island ice cap, arctic Canada. Arctic, 36: 370–371.

Frisch, T., and Morgan, W. C. 1979. Ivory Gull colonies in southeastern Ellesmere Island, Arctic Canada. Canadian Field-Naturalist, 93: 173–174.

Gilchrist, H. G., and Mallory, M. L. 2005a. Population declines and distribution of Ivory Gulls breeding in Canada. Biological Conservation, 121: 303–309.

Haney, J. C., and MacDonald, S. D. 1995. Ivory Gull (*Pagophila eburnea*). In The Birds of North America, No. 175 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Macdonald, S. D., and Macpherson, A. H. 1962. Breeding places of the Ivory Gull in Arctic Canada. National Museum of Canada Bulletin, 183: 111–117.

Mallory, M. L., Gilchrist, H. G., Fontaine, A. J., and Akearok, J. A. 2003. Local ecological knowledge of Ivory Gull declines in Arctic Canada. Arctic, 56: 293–298.

Stenhouse, I. J., Robertson, G. J., and Gilchrist, H. G. 2004. Recoveries and survival rates of Ivory Gulls banded in Nunavut, Canada, 1971–1999. Waterbirds, 27: 486–492.

Thomas, V. G., and MacDonald, S. D. 1987. The breeding distribution and current population status of the Ivory Gull in Canada. Arctic, 40: 211–218.

### 8.2.8 Black-legged Kittiwake (*Rissa tridactyla*)

*Evidence for sensitivity*

There is no indication where the estimate of generation length comes from. Furness and Tasker (2000) should be cited as evidence for the high sensitivity of kittiwakes to changes in prey abundance. It is <u>not</u> correct that the fishery in the 'sandeel box' is managed on the basis of kittiwake 'trends'. A suggestion was made that the fishery should be closed and opened on the basis of threshold levels of kittiwake breeding productivity, but this suggestion has not been adopted by the European Commission. The Commission regularly reviews the closure (originally put in place in 2000) and has until now extended it, but criteria for this extension are not made public.

*Evidence for threat*

There is substantial published indirect evidence for a link between the observed decline in kittiwakes in the UK and lack of sandeels. Frederiksen *et al.* (2004) showed that both breeding productivity and adult survival of kittiwakes was negatively affected by high sea surface temperatures and by the presence of an industrial sandeel fishery, both factors presumed to affect sandeel abundance. Frederiksen *et al.* (2005) showed that kittiwake colonies in the UK could be grouped into regional clusters with similar patterns of temporal variability in breeding productivity, and that these clusters were consistent with sandeel population structure. Frederiksen *et al.* (2006) showed that the abundance of sandeel larvae was strongly related to plankton abundance, and that seabird (incl. kittiwake) breeding productivity was positively related to the abundance of sandeel larvae in the previous year. Oro and Furness (2002) showed that survival rates of adult kittiwakes in Shetland were affected by sandeel abundance and by great skua predation rates.

- The reference to hunting effects in Greenland and the Burnham *et al.* (2006) paper is irrelevant here, as W Greenland is outside the OSPAR area.

- Capelin belongs to the main prey during the breeding season (Barrett and Krasnov, 1996).

- White-tailed Eagles are both direct and indirect (via corvids) predators of chicks and major causes of disturbance in colonies in Norway (Barrett and Krasnov, 1996).

*Sufficiency of data*

There is substantial information about the link between sandeels and kittiwakes, see above …

*Natural variability*

Frederiksen *et al.* (2004) used a population model to predict kittiwake population growth in the North Sea, and showed that if sea temperatures increase further, the observed decline is expected to continue even if the sandeel fishery remains closed.

*Current management*

As described above (Evidence for sensitivity), the statements about fisheries management are incorrect and misleading.

*Required further management*

Again, the remarks about hunting in Greenland are irrelevant here.

*Useful references*

Delete Burnham *et al.* (2006). Change reference to Mitchell *et al.* (2004) to chapter reference (Heubeck, 2004).

Add the following references (as a minimum):

Barrett, R. T., and Krasnov, Y. V. 1996. Recent responses to changes on stocks of prey species by seabirds breeding in the southern Barents Sea. ICES Journal of Marine Science, 53: 713–722.

Frederiksen, M., Wanless, S., Harris, M. P., Rothery, P., and Wilson, L. J. 2004. The role of industrial fisheries and oceanographic change in the decline of North Sea black-legged kittiwakes. Journal of Applied Ecology, 41: 1129–1139.

Frederiksen, M., Wright, P. J., Heubeck, M., Harris, M. P., Mavor, R. A., and Wanless, S. 2005. Regional patterns of kittiwake *Rissa tridactyla* breeding success are related to variability in sandeel recruitment. Marine Ecology Progress Series, 300: 201–211.

Frederiksen, M., Edwards, M., Richardson, A. J., Halliday, N. C., and Wanless, S. 2006. From plankton to top predators: bottom-up control of a marine food web across four trophic levels. Journal of Animal Ecology, 75: 1259–1268.

Furness, R. W., and Tasker, M. L. 2000. Seabird-fishery interactions: quantifying the sensitivity of seabirds to reductions in sandeel abundance, and identification of key areas for sensitive seabirds in the North Sea. Marine Ecology Progress Series, 202: 253–264.

Heubeck, M. 2004. Black-legged Kittiwake *Rissa tridactyla*. Pages 277–290 *in* Mitchell, P. I., Newton, S. F., N. Ratcliffe, and Dunn, T. E., editors. Seabird populations of Britain and Ireland. T. & Poyser, A. D., London.

Oro, D., and Furness, R. W. 2002. Influences of food availability and predation on survival of kittiwakes. Ecology, 83: 2516–2528.

### 8.2.9 Thick-billed Murre = Brünnich's Guillemot (*Uria lomvia*)

*Evidence for sensitivity*

Populations are sensitive to hunting, oil pollution, incidental take in fishing gear and other factors. It is suggested not to characterise this species as having "very low resilience". For example, under extraordinary hunting pressures in Newfoundland in winter, Arctic Canadian populations have managed to maintain themselves and in some cases to increase. A different situation obtains in Greenland where hunting at northern colonies during the summer has caused huge declines. Despite hunting and other pressures, measured apparent survival rates in Canadian and Greenland populations was high (88–90%; Gaston and Hipfner, 2000). Where measured, breeding success is usually high and breeding failure has not been reported. Once breeding has commenced, breeding propensity is high (about 95%). As for the Ivory Gull, it is suggested to add that thick-billed murres are sensitive to climate change and warming in the Arctic (see Gaston *et al.,* 2002).

*Evidence for threat*

1 ) It is not correct that the thick-billed murre hunting in Greenland is unregulated. It has been regulated by national executive order since 1988, and the open season were generally shortened with 3 month in 2002 apparently leading to an about 50% reduction in harvest to a level apparently around 100 000 thick-billed murre annually. (Greenland Home Rule Harvest statistics based on hunters reporting system, Piniarneq, www.nanoq.gl/fangst and F. Merkel pers. com.).

2 ) The effect of the chronic oil pollution off the Newfoundland coast, where also some of the thick-billed murres from Greenland, Norway and Iceland winter, has been assessed by Wiese *et. al.*, 2003, 2004a. Estimates of annual seabird mortality due to this pollution includes 160 000 to 275 000 thick-billed murre. Wiese *et al.* (2004b) estimated that this mortality have reduced population growth in Thick-billed murre colonies in the eastern Canadian Arctic by 2.5%.

*Management considerations*

The review does not mention that the Conservation of Arctic Flora and Fauna (CAFF) program of the Arctic Council drafted an International Murre Conservation Strategy which is being implemented by CAFF member countries, and therefore included countries in the OSPAR region (see references below).

*Useful references.*

CAFF Circumpolar Seabird Working Group. 1996. International murre conservation strategy and action plan. CAFF International Secretariat, Akureyri, Iceland. 16 pp.

Gaston, A. J., Hipfner, J. M., and Campbell, D. 2002. Heat and mosquitoes cause breeding failures and adult mortality in an arctic-nesting seabirds. Ibis, 144: 185–191.

Gaston, A. J., and Hipfner, J. M. 2000. Thick billed Murre (*Uria lomvia*). In The Birds of North America, No. 497 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Wiese, F. K., and Robertson, G. J. 2004. Assessing seabird mortality from chronic oil discharges at sea. Journal of Wildlife Management, 68: 627–638.

Wiese, F. K., Robertson, G. J., and Gaston, A. J. 2004 b. Impacts of chronic marine oil pollution and the murre hunt in Newfoundland on thick-billed murre *Uria lomvia* populations in the eastern Canadian Arctic. Biological Conservation, 116: 205–216.

Wiese, F. K., and Ryan, P. C. 2003. The extent of chronic marine oil pollution in southeastern Newfoundland waters assessed through beached bird surveys 1984–1999. Marine Pollution Bulletin, 46: 1090–1101.

# Annex 1    List of participants

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## Annex 2     WGSE Terms of Reference 2008

The **Working Group on Seabird Ecology** [WGSE] (Chair-elect: Jim Reid, UK) will meet in Lisbon or Madeira, Portugal, 10–14 March 2008 to:

a ) Review the effects of climatic variability (e.g. NAO) and climate changes on seabird distribution, abundance, productivity and life-history parameters in the North Atlantic;

b ) Review the extent to which bycatch may affect seabirds in the North Atlantic;

c ) Review ways in which fisheries management may minimize impacts on seabirds;

d ) Finalise to consider ecological issues linked to the circulation of parasites and pathogens within seabird populations.

WGSE will report by 30 April 2008 to the attention of the Living Resources Committee.

### Supporting Information

| | |
|---|---|
| **PRIORITY:** | This is the only forum for work being carried out by ICES in relation to marine birds. If ICES wishes to maintain its profile in this area of work, then the activities of WGSE must be regarded as of high priority. |
| **SCIENTIFIC JUSTIFICATION AND RELATION TO ACTION PLAN:** | Action Plan No: 1.2, 1.7, 1.8, 2.2, 4.1, 4.15<br><br>Term of Reference a) The review for OSPAR conducted at the 2007 WGSE meeting showed that more work is needed to understand the effects of both climatic variability and climate changes on seabirds.<br><br>Term of Reference b) WGSE thinks that it is timely to review this issue for the North Atlantic. Most concerns have so far related to the Southern Ocean but it is probable that bycatch in the coastal and offshore waters of the North Atlantic may be of concern for some seabirds.<br><br>Term of Reference c) WGSE suggests to summarize and evaluate progress made on this topic, and to provide ICES with suggestions on how to further improve ecosystem effects of fishing activities.<br><br>Term of Reference d) This comprehensive TOR could not be finalised at the 2006 WGSE meeting. |
| **RESOURCE REQUIREMENTS:** | Facilities for WGSE to work in Madeira are anticipated to be excellent. |
| **PARTICIPANTS:** | The Group is normally attended by ca. 15 members and guests. The Working Group should be able to achieve most the above objectives. However, some members may not be able to attend through lack of funding. Funding of these members from Member Countries would be very welcome. |
| **SECRETARIAT FACILITIES:** | None. |
| **FINANCIAL:** | No financial implications. |
| **LINKAGES TO ADVISORY COMMITTEES:** | ACE, ACME |
| **LINKAGES TO OTHER COMMITTEES OR GROUPS:** | WGSE is keen to continue the process of integration of seabird ecology into ICES work. |
| **LINKAGES TO OTHER ORGANIZATIONS:** | EU, OSPAR, HELCOM |

## Annex 3 English and scientific names of birds mentioned in this report

| ENGLISH NAME | SCIENTIFIC NAME |
|---|---|
| Red-throated diver | *Gavia stellata* |
| Black-throated diver | *Gavia arctica* |
| Slavonian grebe | *Podiceps auritus* |
| Great crested grebe | *Podiceps cristatus* |
| Red-necked grebe | *Podiceps grisegena* |
| Northern fulmar | *Fulmarus glacialis* |
| Cory's shearwater | *Calonectris diomedea* |
| Westland petrel | *Procellaria westlandica* |
| White-chinned petrel | *Proceallaria aequinoctialis* |
| Little shearwater | *Puffinus assimilis* |
| Balearic shearwater | *Puffinus mauretanicus* |
| Manx shearwater | *Puffinus puffinus* |
| Bulwer's petrel | *Bulweria bulwerii* |
| Band rumped = Madeiran storm-petrel | *Oceanodroma castro* |
| Leach's storm-petrel | *Oceanodroma leucorhoa* |
| Northern gannet | *Morus bassanus* |
| Great cormorant | *Phalacrocorax carbo* |
| European shag | *Phalacrocorax aristotelis* |
| Eurasian wigeon | *Anas penelope* |
| Greater scaup | *Aythya marila* |
| Common eider | *Somateria mollissima* |
| Long-tailed duck | *Clangula hyemalis* |
| Common scoter | *Melanitta nigra* |
| Velvet scoter | *Melanitta fusca* |
| Red-necked phalarope | *Phalaropus lobatus* |
| Arctic skua | *Stercorarius parasiticus* |
| Great skua | *Stercorarius skua* |
| Little gull | *Larus minutus* |
| Audouin's gull | *Larus audouinii* |
| Mediterranean gull | *Larus melanocephalus* |
| Common gull | *Larus canus* |
| Herring gull | *Larus argentatus* |
| Lesser black-backed gull | *Larus fuiscus* |
| Greater black-backed gull | *Larus marinus* |
| Ivory gull | *Pagophila eburnea* |
| Black-legged kittiwake | *Rissa tridactyla* |
| Roseate tern | *Sterna dougallii* |
| Common tern | *Sterna hirundo* |
| Little tern | *Sterna albifrons* |
| Sandwich tern | *Sterna sandvicensis* |
| Gull-billed tern | *Gelochelidon nilotica* |
| Common guillemot | *Uria aalge* |
| Razorbill | *Alca torda* |
| Black guillemot | *Cepphus grylle* |
| Brünnich's guillemot | *Uria lomvia* |
| Little auk | *Alle alle* |
| Atlantic puffin | *Fratercula arctica* |